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The Environmental Impacts of Agriculture: A Review

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ABSTRACT

Agriculture and the entire food production system play a critical role in sustaining the human species. However, as we strive to secure our means of subsistence, our extensive use of land and water has led to the depletion of the environment and biodiversity. This raises the pressing question of whether we can sufficiently produce food to support a growing population while simultaneously mitigating the inevitable environmental impacts. This article presents a comprehensive review of the significant effects of agriculture on the environment including contributions to greenhouse emissions, land use and land-use change, and forests, impact on biodiversity, impact on water quality and quantity, and the impact of pesticide use. The article also offers a list of approaches used to measure and evaluate the impact of agriculture. The primary aim of this article is to comprehensively review the latest insights in the field and to stimulate further research in this area.

Keywords: Agriculture; impact; environment; mitigation; adaptation; assessment methods; climate; change

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1 Introduction

Agricultural activity[1] impacts the environment, and environmental disruptions affect agriculture activities. Over time, it has been noted that agriculture plays a crucial role in initiating economic development, but as a country attains a certain level of development, agriculture tends to diminish in significance within the national economy (Byerlee *et al.*, 2009). Although, agriculture accounts for a small share of the global economy; it is central to the livelihood of many people (Alston and Pardey, 2014). In 2019, 27% of the total employment in the world was in agriculture, and only 4% of the world's gross domestic product (GDP) was in the agriculture (World Bank, 2021). The contribution of agriculture to countries' GDP varies. For middle- and low-income countries, as classified by the World Bank (2022),[2] there is a larger share of employment in agriculture, and agriculture contributes a larger share to the country's GDP, than high-income countries (Alston and Pardey, 2014).

In the late twentieth century, large-scale and well-financed agricultural practices became associated with environmental issues such as the deterioration of traditional landscapes, the pollution of waterways through eutrophication, and soil erosion. These issues have a notable impact on ecosystems and biodiversity (Bryant *et al.*, 2020; Rosegrant *et al.*, 2009). Scientists highlight that in the nineteenth and early twentieth centuries, the primary objective of agriculture was to boost productivity rather than focusing on sustainability. Currently, the emphasis has shifted toward improving agricultural practices to address the growing global population's food needs while also meeting sustainability objectives (Lamichhane, 2017). However, the controversy

[1]The scope of the word "Agriculture" is varied. Agriculture refers to the farming practice itself, including crops and livestock. Agriculture could also include land use and land-use change which has forestry associations. Finally, agriculture could include the entire food system, the food supply chain, input supply, farming, processing, shipping, storage, and retailing. In the manuscript, we will make clear what dimension of agriculture we are referring to.

[2]The designation of middle- and low-income countries are after the World Bank classification. In 2022, low-income countries exhibited an annual per capita income of less than $1,085; lower-middle income ranged between $1,086–4,255; upper-middle-income ranged between $4,256 and $13,205; and high income was greater than $13,205.

arises as some scientists also argue that the issue of food security is more related to food distribution and efficient use, more than increasing the total amounts of food to be produced. In fact, the U.N. Food and Agriculture Organization, reports that in 2021 approximately one-third of the globally produced food was wasted (U.N. Food and Agriculture Organization, 2021).

The relationship agriculture environment is bidirectional, while agricultural practices impact the environment, changes in climate patterns also impact agriculture. According to the Intergovernmental Panel on Climate Change (IPCC, 2019), the increasing temperatures and shifting rainfall patterns, coupled with a higher frequency of extreme weather events, are projected to have adverse impacts on the production of essential food staples like maize, wheat, livestock, and fish/seafood. The well-being of crop cultivation and livestock farming is closely intertwined with the health of ecosystems. The current and future status of natural resource, requirements, and limitations in the agriculture sector are profoundly influenced by environmental conditions (U.N. Food and Agriculture Organization, 2018). Elevated atmospheric carbon dioxide levels are expected to result in reductions in the zinc, iron, and protein concentrations found in fundamental foodstuffs like wheat, rice, peas, and soybeans (U.N. Food and Agriculture Organization, 2016). Rising temperatures are influencing various aspects of the marine environment, including sea-surface temperatures, ocean circulation, wave patterns, storm systems, salinity levels, oxygen concentrations, and acidity. These changes are having an impact on fish populations and marine life (U.N. Food and Agriculture Organization, 2018). Elevated temperatures will also contribute to a less dependable freshwater supply, which will pose challenges for small-scale livestock farming, particularly in arid and semi-arid grassland and rangeland ecosystems. The combination of higher temperatures and diminished water availability will have adverse effects on animal well-being, that some scientists believe will ultimately reduce the quality and quantity of available feed and fodder resources (U.N. Food and Agriculture Organization, 2018).

The impacts of climate change vary among nations and different social segments within a nation, leading to shifts in income distribution, the availability of natural resources, and income-earning prospects. Typically, it is the most economically disadvantaged segments of the population, often reliant on ecosystem services, that bear the consequences from the degradation of ecosystems (Rosegrant *et al.*, 2009). Regions characterized by elevated temperatures, degraded landscapes, and limited adaptation capabilities, often found in middle- and low-income countries, are likely to experience more pronounced impacts (Mendelsohn and Dinar, 2009). The susceptibility of livelihoods to climate change, as measured by factors like the proportion of income of GDP derived from agriculture, forestry, and fishing, as well as the capacity for adaptation, will play a crucial role in determining the impact of climate change on different regions and communities (Van Vuuren *et al.*, 2011).

In view of the critical role agriculture and food production play in sustaining humanity and the environmental challenges current practices pose, one questions if it is possible to balance food production and environmental preservation. Literature reviewed concludes that to achieve this balance, it is important to explore alternative approaches, integrating new technologies, and implementing effective policies, especially in developing countries. This article offers a comprehensive review of how agricultural practices affect the environment and presents various methodologies for assessing this impact. By doing so, the aim is to contribute to the identification of effective adaptation and mitigation strategies, technologies, and policies. The article is organized as follows: the next sections include agricultural impacts to greenhouse gasses (GHG) emissions, land use and land-use change and forests, agriculture's impact on biodiversity and wild/nondomesticated species, impact on water quantity and quality, and pesticides impact on the environment. Considering the importance of providing accurate estimates to implement effective measures the review also includes a discussion of selected approaches to assess agriculture's environmental impacts. Finally, the review includes a discussion on agricultural adaptation and conclusions.

2 Agricultural Contributions to GHG Emissions

One of the significant impacts of agriculture is the alteration of the atmosphere's composition by increasing GHG such as methane, carbon dioxide, nitrous oxide, and fluorinated gases (Kross *et al.*, 2022; Turcotte *et al.*, 2017; U.N. Food and Agriculture Organization, 2020). This atmosphere's alteration increased the average global temperatures and is predicted to rise by 2°C by 2100 (Malhi *et al.*, 2021). In 2018, more than half of total nitrogen emissions came from food systems (encompasses food production, processing, packaging, transport, retail, consumption, and disposal) (Crippa *et al.*, 2022). This section offers a review of GHG emissions from crop and livestock farming including agricultural soil management, livestock enteric fermentation, manure management, and rice cultivation.

In 2015, the top six emitting countries absorbed 51% of all the global food system emissions: China (13.5%), Indonesia (8.8%), the United States (8.2%), Brazil (7.4%), EU-28 (6.7%), and India (6.3%). The total share of the food systems to total GHG emissions decreased over time in developing countries from 68% in 1990 to 39% in 2015, except for China, where emissions grew by 41%. Industrialized countries have maintained a stable 24% of emissions (Crippa *et al.*, 2021). The decrease in the share of food system GHG emissions over the years is attributed to the increases in nonfood emissions and reductions in the deforestation (Crippa *et al.*, 2021; Tubiello *et al.*, 2015). Further, the share of the specific agricultural activities (crop and livestock) to the total GHG emissions varies across countries. The accurate assessment of the contributions of agriculture to GHG emissions is of critical importance for countries were agriculture represents a larger share of the economy, so climate change strategies can be implemented through effective policies and sufficient funding (U.N. Food and Agriculture Organization, 2014).

Globally, Asia is the largest contributor to agricultural GHG emissions absorbing 44% of total global emissions. The Americas is in the second place with 26%, followed by Africa (15%), and Europe (12%) (U.N. Food and Agriculture Organization, 2014). Worldwide, enteric fermentation is the largest contributor to agricultural GHG emissions with 40%, followed by manure left on pasture (16%), synthetic fertilizers (13%), rice cultivation (10%), manure management (7%) and burning of savannas (5%) (U.N. Food and Agriculture Organization, 2014). Just for comparison purposes, the U.S. Environmental Protection Agency (2022) reports that in 2021 in the United States, agricultural soil management (nitrogen applied to soils) represented 49% of the total GHG emissions, followed by enteric fermentation (33%), manure management

(14%), rice cultivation (3%), urea and liming application (1.4%), and the field burning of residues (0.12%).[3]

Among the volume of gases, methane accounts for 35% of all food-system GHG emissions in both developing and industrialized countries. Livestock production, farming, and waste treatment are the primary sources of methane. At the same time, rice leads the food crops' methane emissions. The nitrous oxide emissions are comparable between developing and industrialized countries. The increase in food supply-chain emissions is more prevalent in industrialized countries. For example, fluorinated gas emissions are more prevalent in industrialized countries, as they are linked to refrigeration at food retailing. Also, the share of carbon dioxide for energy is more prominent in developing countries. This reflects that agriculture in these developing countries has become more mechanized, along with higher uses of fertilizers and pesticides (Crippa *et al.*, 2021).

The accurate measurement of the GHG emission of the different stages of the food supply chain is essential to identify impactful policy actions to mitigate GHG from the food system. For example, it is reported that food packaging contributes more to GHG emissions than food distribution in the different food supply-chain stages. It is clear that agriculture and, in general, the food system has increased its energy use in industrialized and is increasing it in developing countries. Therefore, policies directed to increase energy efficiency and decarbonization are warranted (Crippa *et al.*, 2021).

2.1 Soil Management

Applying synthetic nitrogen fertilizers and cultivating nitrogen-fixing crops are essential for global agriculture's economic sustainability

[3]The U.S. Environmental Protection Agency (2021) reports that the emissions are calculated using internationally accepted methods provided by the Intergovernmental Panel on Climate Change (IPCC) following the 2006 IPCC Guidelines for National Greenhouse Gas Inventories. *Source*: https://www.ipcc.ch/report/2006-ipcc-guidelines-for-national-greenhouse-gas-inventories/.

The reporting format is consistent with the United Nations Framework Convention on Climate Change (UNFCC) guidelines. *Source*: https://unfccc.int/process-and-meetings/transparency-and-reporting/reporting-and-review-under-the-convention/greenhouse-gas-inventories-annex-i-parties/reporting-requirements?gclid=CjwKCAjw6eWnBhAKEiwADpnw9pChL6h9ErUFm0Qpf0q0bDDI__J0l6j9pxMDz39VlAS3piillqK_KhoCffMQAvD_BwE.

(Cassman and Dobermann, 2022; Fowler *et al.*, 2013; Sobota *et al.*, 2015). The increased use of nitrogen fertilizer is critical for grain production (Zhao *et al.*, 2021). The Food and Agriculture Organization (FAO) projects that by 2050, the human population will reach 10 billion (in 2022, it reached 8 billion), and an additional 52 million tons of nitrogen fertilizer in conjunction with 165 million hectares of new arable land will be required to meet the demand for food, feed, fiber, and biofuel (U.N. Food and Agriculture Organization, 2018). Scientists coincide in that the application of synthetic fertilizers must be evaluated to be prepared for the unprecedented demand for food and mitigate negative impacts on society and the environment (Rauh, 2021).

The overuse and unintended nitrogen leakage pose significant health and environmental concerns. The most widespread, costly, and challenging environmental problems caused by excess nitrogen in water and air include increased mortality and morbidity due to nitrous oxide contamination in both the air and drinking water. Additionally, environmental hazards include water and soil acidification, pollution of groundwater and other mineral water resources, as well as the acceleration of the ozone layer degradation (Bashir *et al.*, 2013). Another effect of nitrogen leaching and run-off from agricultural soils is the increased frequency and severity of toxic algae blooms and hypoxia (death) in freshwater and coastal marine ecosystems (Sobota *et al.*, 2015).

The issue of excessive usage is particularly critical, especially when considering that, the majority of cereal crops only need about half of the nitrogen fertilizer that is typically applied (Zhao *et al.*, 2021). Although, other studies noted that it is difficult to precisely estimate the amount of nitrogen to be applied to meet the crop's physiological need (Cassman and Dobermann, 2022).

Scientists have observed a complex relationship between climate, nitrogen pollution, and the emergence of hypoxic zones. On one side, rising temperatures adversely affect crop yields. In response to reduced yields, farmers compensate by increasing fertilizer application, or alternatively the plant might absorb less nitrogen, resulting in a surplus remaining in the soil that subsequently leaches into nearby watersheds (Metaxoglou and Aaron, 2022). In large parts of the world, especially in sub-Saharan Africa, soils experience nitrogen deficiency and low yields due to insufficient applications of nitrogen and other nutrients (Cassman and Dobermann, 2022).

In the United States nitrogen leakage intensity is near twice the global average (Sobota *et al.*, 2015). The application of synthetic nitrogen fertilizers cultivated biological nitrogen fixation by soybeans and alfalfa and confined animal feeding operations manure are prevalent in the nitrogen loading and leakage. The areas with the most considerable nitrogen inputs had the highest damage costs: upper Midwest and Central California (Sobota *et al.*, 2015). Nitrogen from row crops, large farms, and concentrated animal feeding operations run off the Mississippi River Basin. The nitrogen exported from the Mississippi River Basin to the Gulf of Mexico has created a zone with very low oxygen levels, jeopardizing fish and other marine life. This zone is the second largest hypoxic in the world, only behind the dead zone of the Arabian sea (Metaxoglou and Aaron, 2022). On top, nutrient pollution increases turf and macroalgal cover, killing coral reefs in marine systems (Zhao *et al.*, 2021).

Quantifying the social cost of nitrogen leakage is challenging because there are multiple loss pathways and endpoints at which damages occur (Metaxoglou and Aaron, 2022). Studies estimated that nearly 71% of the anthropogenic nitrogen leaked to the environment is in water resources. The costs associated with nitrogen leakage to the environment in the United States added to $210 billion ($81–441 billion) annually in the early 2000s. Similar ranges were estimated for the European Union at $97–625 billion (Sobota *et al.*, 2015). In China, these costs were estimated at about $91 billion (Yu *et al.*, 2019).

Alternative strategies to improve nutrient management to reduce nitrogen leakage include the breeding of new varieties of crops that would require less nitrogen (Sobota *et al.*, 2015). Because the problem is the excessive use of nitrogen, scientists suggest careful management in nitrogen application, applying practices like the 4Rs in nutrient management: right source, rate, time, and place for nutrient application (Metaxoglou and Aaron, 2022). On-site control of nitrogen application, besides controlling application rates and timing of nitrogen application, includes better management of manure spreading, nitrification inhibitors, change of tillage methods, and increasing drainage tile spacing. Another approach is constructing and restoring riparian zones and wetlands that could act as buffer systems between the agricultural land and the watershed (Metaxoglou and Aaron, 2022). A study conducted in China revealed that combining synthetic fertilizers with livestock manure could improve soil quality and boost crop productivity to a greater extent compared to the use of the synthetic fertilizer alone (Yu *et al.*, 2019). Nutrient recycling shows promise in reducing the need for synthetic nitrogen applications. The study identified three recycling strategies: traditional wet manure recycling, dry compost recycling, and direct wastewater recycling. The associated costs included an initial investment of approximately US $104 billion and annual operational costs, at US $29 billion (Yu *et al.*, 2019).

2.2 Livestock Enteric Fermentation

Ruminant livestock enteric fermentation and manure management significantly contribute to GHG emissions. Cattle animals experience rumen microbial fermentation, also known as enteric fermentation, which generates a series of gases, such as carbon dioxide and methane, exhaled or eructated by the animal. Enteric methane is produced in the anaerobic conditions of the animal's rumen. The process involves the methanogenic Archaea microorganisms that utilizes carbon dioxide and hydrogen to produce methane (Eckard *et al.*, 2010). The eructation of gases prevents the animal's bloating and is the primary route for methane emission to the atmosphere. Greater feed consumption volumes and diminished feed quality are directly linked to increased methane emissions. The size of the animal, its growth rate, activity levels, and its ultimate purpose (whether for milk or meat production), along with its growth stage, and pregnancy conditions, all have an impact on methane emissions. Researchers have noted that enteric fermentation decreases the intake of digestible energy, which could otherwise be allocated to functions such as growth or milk production. Consequently, reducing methane emissions from enteric fermentation benefits the environment and enhances the economic viability of cattle operations (Min *et al.*, 2022).

There are several mitigation strategies for cattle emission of methane, mostly applied in North America and Europe. These strategies include dietary/nutritional, reproductive, genetic, and management interventions (Eckard *et al.*, 2010). Dietary interventions influence the rumen microbiome affecting the rumen fermentation profiles and microbiota community to reduce GHG emissions (Min *et al.*, 2022). In the United States, the use of feed additives such as 3-nitroxypropanol (3-NOP) for methane inhibition shows promise and to be more effective with dairy than with beef cattle (Dillon *et al.*, 2021). Studies in Europe confirmed that this additive is critical for reductions in enteric fermentation (Aan den Toorn *et al.*, 2021). Other studies advocate using plant-based products such as condensed tannins and saponins as methane inhibitors. Others recommend the use of phytochemicals with anthelmintic and antioxidant properties. Further, essential oils such as oregano and thyme have successfully reduced methane in vitro experiments, but the in vivo trials still need to be completed. In addition, seaweed (*Asparagopsis taxiformis*) was proven to reduce emissions (Dillon *et al.*, 2021).

Genetic selection also shows as a promising alternative. Scientists found that methane emissions from livestock are moderately heritable. Therefore, a choice for trait improvement is possible. Scientists recommend a combination of practices like genetic selection and management decisions, such as forage characteristics. Moreover, selection programs to improve cattle feed efficiency are balanced with other outcomes, such as longevity. And finally, the use of gene editing shows promise. Scientists highlight the importance of considering genetic selection in the animal's living environment to achieve optimal productivity (Dillon *et al.*, 2021). Yet more research is needed to identify the optimal strategy to reduce GHG emissions and increase ruminant production efficiency.

There is the need to enhance the accuracy of methane emission measurement Worldwide: China (Tang *et al.*, 2019), South Korea (Ibidhi *et al.*, 2021), Nepal (Thakuri *et al.*, 2020), South Africa (Tongwane and Moeletsi, 2020), Turkey (Kumaş and Akyüz, 2023), and Mexico (Rivera-Huerta *et al.*, 2022). In many countries GHG measurements are inaccurate or inexistent. Enhancements in the accuracy of these measurements are important for evaluating and assessing the effectiveness of mitigation actions to reduce methane emissions. In China, methane emissions resulting from enteric fermentation from grassland cattle have not been adequately evaluated. Research indicates that transitioning from intensive to sustainable grazing practices enhance grassland and livestock production, while offering potential to reduce methane emissions from grazing cattle (Tang *et al.*, 2019). In South Korea, initiatives aimed to enhance the precision of methane measurement for dairy cattle revealed correlations with factors such as feed digestibility, milk production levels, and methane conversion rates (Ibidhi *et al.*, 2021). In Nepal, improvements in feeding composition, technological improvements, and carbon offset mechanisms contribute to efforts to balance GHG emissions (Thakuri *et al.*, 2020). In South Africa, dairy cattle exhibit the most significant emissions, followed by subsistence cattle and commercial beef cattle. Emission factors for commercial beef and dairy cattle in this country surpass those in other African regions but align with figures observed in Europe and North America (Tongwane and Moeletsi, 2020). In Turkey, a study carried in the Lakes region found that livestock complements other agricultural activities. Therefore, policies aimed to mitigate GHG emissions should consider the structure of the entire industry (Kumaş and Akyüz, 2023). In Mexico, efforts to reduce GHG emissions from livestock include management of livestock feed. This implies regulating the quality and composition of animal diets, which directly impact how efficiently the animals digest their feed. Grazing livestock is prevalent in Mexico, therefore studies on digestibility particularly for livestock that primarily consume native vegetation could provide insights into improving feed management practices to reduce emissions (Rivera-Huerta *et al.*, 2022).

2.3 Manure Management

Manure management involves the handling, disposal, and storage of livestock waste (feces and urine) with the aim of conserving and reutilizing valuable nutrients. Animal manure contains a rich set of nutrients required for plant growth and could represent a significant source for nitrogen for intensive and subsistence crop systems (Montes *et al.*, 2013).

In the United States, the shift from smaller, pasture-based farming operations to large, concentrated animal feeding operations has resulted in the concentrated accumulation of excessive volumes of manure in a particular location. This concentration of manure has become an environmental issue (Malomo *et al.*, 2018; Rauh, 2021). The manure contains microorganisms that could harm humans and animals, causing food contamination and threatening public health (Malomo *et al.*, 2018).

As previously described, the manure also contains useful organic materials that could be recycled under careful management. The uses include soil nutrients, compost, fertilizer, organic matter as soil amendments/structuring and soils such as bedding. Moreover, manure can be used as a source of energy, such as biogas and bio-oil, and fiber as a peat substitute, paper, and to build materials (Malomo *et al.*, 2018).

Studies proved that the beneficial reuse of these organic materials, under careful management, could increase soil organic content and water holding capacity over time. However, this reuse is subject to management cost, access to a viable market, variable nutrient composition, risk, public acceptance, and regulation compliance (Rauh, 2021). Due to the interconnections among the animal care, storage, and land application stages of manure management, it is imperative to approach manure management as an integrated part of the overall livestock production system rather than as an isolated practice (Montes *et al.*, 2013).

Research conducted in Europe showed that the separation of slurry has led to a reduction in GHG emissions, and that the combination of slurry separation with incineration has proven even more effective in decreasing these emissions (Sommer *et al.*, 2009). Research in the United States identified significative differences in manure handling between large- and small-scale dairy farms. Small farms handle solid manure and apply to the land daily resulting in lower GHG emissions than large farms. In contrast, large farms can handle liquid manure, implement long-term storage, and invest in mitigating practices such as sand separation, solid–liquid separation, and anaerobic digestion. The storage of liquid manure without undergoing processing can contribute the most to GHG emissions, whereas the implementation of manure processing through anaerobic digestion significantly reduces these emissions (Aguirre-Villegas and Larson, 2017). Studies in China indicates that approaches to reduce GHG emissions must efficiently combine crop and animal production. For example, during the manure storage phase, the use of dry collection technologies, acidification, compaction, mulching, as well as the utilization of biosolids and biochar, have proven to be efficient in reducing GHG emissions (Zhang *et al.*, 2023). Studies conducted in Australia support the notion that it is essential to comprehensively grasp the entire context of agricultural practices before advocating abatement strategies to stakeholders. Many of these abatement options are better suited for intensive animal production systems, with fewer viable choices available for extensive grazing systems (Eckard *et al.*, 2010).

Anticipated technological advancements are poised to play a pivotal role in enhancing manure management practices. The absence of sustainable strategies, both environmentally and economically, will inevitably lead to environmental pollution (Malomo *et al.*, 2018). In this context, it is crucial to develop more accurate models for estimating emissions at various scales, from individual farms to national levels. Research indicates that GHG emissions from manure management exhibit variability across different farming systems, and depend on climate, systems design, and management practices (Sommer *et al.*, 2009).

2.4 Rice Cultivation

Rice is a staple for half of the global population and its demand in future years is likely to increase. Cultivation practices typically involve flooding a rice field. This suppresses the oxygen supply from the atmosphere to the soil, leading to anaerobic fermentation of soil organic matter (Neue, 1993b). The process consists of two parts. First, fermentative bacteria hydrolyze complex organic compounds such as polysaccharides, proteins, and neutral fats to carbon dioxide, hydrogen and acetate. Second, the latter gases are reduced to methane by methanogenic microbes, or methanogens (Thauer *et al.*, 2008). Methane is released from the submerged soils to the atmosphere by diffusion and ebullition and through the plant's roots and stems (Neue, 1993a). Emissions from rice fields are influenced by the type of farming: irrigated, rainfed, or deep water, if nitrogen fertilizer is used, the organic input, and rice varieties (Zhang *et al.*, 2016).

Seventy-eight percent of the methane originated from rice is concentrated in irrigated areas comprising 60% of total rice harvested area. India takes the lead with 27% of the global methane from rice, followed by China with 23%. Vietnam rice production consisting of flooding and triple cropping and contributes to 10% of global rice methane emissions (Carlson *et al.*, 2017). Measuring rice contributions to the GHG emissions is crucial; however, the environmental complexity of rice production challenges such measurements. For example, increased temperatures lead to root decay fostering methane production. Less precipitation in rainfed rice leads to methane emissions reductions (Zhang *et al.*, 2016).

There is a diverse set of strategies to mitigate rice emission of methane. These include different irrigation methods such as midseason drainage, intermittent irrigation, alternative wetting and drying, direct dry seeding and aerobic rice cultivation. Also, recent studies demonstrated that drip irrigation can mitigate methane emission (Parthasarathi *et al.*, 2019). The application of nitrogen fertilizer also affects GHG emissions. However, there is no consensus on the net impacts of nitrogen fertilizers on these emissions, and the effect likely depends on site specific factors such as the type of nitrogen fertilizer and water management. In principle, nitrogen fertilizers stimulate crop growth and provide more carbon substrates to methanogens to produce methane. On the type of nitrogen fertilizer, the stimulation of methanogens by nitrogen depends on the chemical type of nitrogen. If the nitrogen fertilizer is urea, then the methane emission is greater compared to ammonium sulfate. Water management impacts the net effects of nitrogen fertilizers on the methane emission, favoring intermittent drainage water compared to flood irrigation (Banger *et al.*, 2012).

3 Land Use, Land-Use Change, and Forests

According to the Intergovernmental Panel on Climate Change (IPCC), approximately one-quarter to one-third of the Earth's potential net primary production is dedicated to various purposes, including food, feed, fiber, timber, and energy production. Land serves as the foundation for numerous ecosystem functions and services, encompassing cultural and regulatory roles. The annual value of these services has been estimated to be equivalent to the global yearly GDP (IPCC, 2019).

Human activities have significantly transformed the worldwide landscape, leading to the conversion of forests and grasslands into croplands, pastures, and urban areas. Land use, land-use change and forestry (LULUCF) is the anthropogenic change of land use from one type to another, for example, from forest to crops (Perminova *et al.*, 2016).

According to the World Bank (2016), forested areas have experienced a 3% decline over the past 25 years, primarily due to the expansion of agriculture. Despite these declines, approximately 30% of the Earth's land area still comprises forests (Kim *et al.*, 2017).

Forests hold direct value for humans as they serve as sources of timber, various plant and animal resources, tourist attractions, and recreational spaces. Additionally, they offer indirect benefits by providing watershed protection, which ensures a clean and stable water supply, and by acting as crucial carbon sinks aiding in the storage of carbon and mitigating climate change (Kim *et al.*, 2017). Conversely, forest degradation pertains to the deterioration of land within forests, while deforestation involves the transformation of forested areas into nonforest land, leading to land degradation (IPCC, 2019).

Forests play a dual role in the context of GHG by serving both as a source and a sink, while playing a crucial role in the exchange of energy, water, and aerosols between the Earth's surface and the atmosphere (IPCC, 2019). The conversion of many older secondary and mature old-growth forests into cropland has led to a reduction in the carbon stored on the ground and contributed to an increase in atmospheric carbon levels. In this sense, forestry activities can be considered as either carbon neutral or a net carbon sink, depending on whether they involve carbon removal through forest regrowth or carbon release through deforestation or timber harvesting (Mendelsohn and Dinar, 2009; Sohngen, 2020). The sink effect refers to the capacity to achieve negative GHG emissions which would occur if positive emissions from deforestation and wood harvest were eliminated. Over the past century, forests have acted as net carbon sink, absorbing more carbon than they release (Sohngen, 2020). The relatively low carbon stock per hectare in many forests presents an opportunity to incentivize forest owners to increase the storage of carbon in forests via carbon sequestration program rather than releasing into the atmosphere (Mendelsohn and Dinar, 2009).

Climate change, marked by raising concentrations of carbon dioxide in the atmosphere and escalating temperatures, is expected to have significant impacts on forests. These effects will likely include changes in the growth rates of trees, shifts in the disturbance patterns such as wildfires or insect outbreaks, and alterations in the optimal geographic locations where certain tree species can thrive. For instance, scientists have noted that climate change has already influenced the migration patterns of forests in North American, highlighting its substantial impact on the ecosystems (Sohngen, 2020). As such, the anticipated global warming is expected to alter the mix of land-use practices in various regions around the world. That is, the types of activities and land uses that are most suitable in certain areas might shift as a result of the changing climate and its impact on ecosystems (Mendelsohn and Dinar, 2009). Furthermore, climate change amplifies the land degradation by increasing the intensity of rainfall, causing more frequent and severe flooding and drought events, heat stress, extended dry periods, strong winds, rising sea levels, and more vigorous wave action. Coastal erosion is an ongoing issue that is escalating and encroaching upon additional regions, and the elevation of sea levels is compounding land-use challenges in certain areas around the globe (IPCC, 2019).

Deforestation has contributed to significant share of global GHG emissions. Note that about one-third of the total food-system GHG emissions are originated from land use and land-use-change activities.[4] These emissions are composed of carbon losses from deforestation and the degradation of organic soils (Crippa *et al.*, 2021). Halting deforestation and allowing secondary forests to naturally regenerate has the potential to yield cumulative negative emissions between 2016 and 2100 (Houghton and Nassikas, 2018). Studies have demonstrated that the process of converting old-growth forests into secondary-cut forests have resulted in a modest improvement in carbon emissions. This is because the old forests have grown approximately as much as they have decayed, meaning they did not absorb a substantial amount of carbon from the atmosphere. Conversely as secondary forests continue to grow, they serve as a significant carbon sink (Mendelsohn and Dinar, 2009).

LULUCF contributions to climate change will be mitigated by preserving and enhancing the capacity of ecosystems to store carbon, effectively turning them into carbon sinks, and reducing GHG emissions

[4]Note that authors Crippa *et al.* (2021) and Tubiello *et al.* (2015) encompass land use and land-use change and forestry (LULUF or LULUC) as part of the food system. Agencies like the U.S. Environmental Protection Agency refer to farming only when reporting agriculture contributions to emissions, and LULUF/LULUC is considered apart.

resulting from deforestation (Grassi *et al.*, 2017). Harvesting wood products plays a crucial role in the carbon cycle between forests and the atmosphere. While the utilization of harvested wood products can store carbon for varying durations, it also has the potential to replace GHG-intensive materials such as fossil fuels with wood-based energy (Geng *et al.*, 2017). Research findings suggest that when it comes to reducing GHG emissions, replacing non-wood with wood product is more effective than substituting wood for fossil fuels. However, it is important to note that harvest wood products and bioenergy sourced from sustainably managed forests can make a significant long-term contribution to GHG reductions (Geng *et al.*, 2017).

Analyzing the contributions of LULUCF to climate change, across countries, is complex. The integration of forests into international climate change agreements has proved to be intricate (Grassi *et al.*, 2017). For the 2015 International climate negotiations in Paris, France, countries worldwide submitted their Intended Nationally Determined Contributions (INDCs) to the United Nations Framework Convention on Climate Change (UNFCCC). These contributions outlined the specific actions and commitments each country intended to take to address climate change and the inclusion from LULUCF activities was not homogeneous across countries (Forsell *et al.*, 2016). Another critical concern is the measurement of carbon emissions resulting from deforestation. Thanks to advancements in satellite and remote sensing technologies, it is possible to monitor the forest cover across the entire planet. This technological progress has significantly enhanced the ability to assess more accurately the carbon impact of deforestation (Mendelsohn *et al.*, 2016).

Studies conclude that if effective management practices are put into action (e.g., expansion of forest area, greater efficiencies in converting harvested wood to long-lasting products and sources of energy, and novel approaches to sequester carbon in soils) it is expected that the net LULUCF emissions will decrease by 2030 when compared to the levels observed in 2010 (Forsell *et al.*, 2016; Grassi *et al.*, 2017). The most significant reductions are projected to come from Indonesia, Brazil, China, and Ethiopia (Forsell *et al.*, 2016).

Improving reporting accuracy and achieving greater harmonization in LULUCF–GHG emissions data across countries are critical steps for effective mitigation actions. For instance, projections of emissions suggest that certain regions have forests acting as net annual carbon sinks, while others not. Research indicates that within the EU-28 countries, the LULUCF sink offsets approximately 7% of the total EU-28 GHG emissions. However, this percentage varies significantly among member states; for instance, the offset in the Netherlands is minimal, and in Latvia, it is considerably higher (Blujdea *et al.*, 2015). Interestingly, land-use changes constitute roughly 10% of the entire EU-28 land area but contribute to around 20% of the total annual net LULUCF sink. Whereas, organic soils used in cropland make up approximately 9% of the total land area but contribute to over 25% of the total sink. Lastly, forest fires, which emit carbon, account for an average of 3% of the total sink (Blujdea *et al.*, 2015).

When calculating GHG emissions from LULUCF, it is crucial for researchers to consider landscape fires, given their frequent occurrence. A comprehensive approach to carbon emission accounting would offer stronger incentives to enhance fire management practices aimed at reducing the frequency, severity, and extent of uncontrolled landscape fires (Bowman *et al.*, 2023). Specific areas requiring improvement in national inventories were identified, particularly in Australia. These improvements involve focusing on forest fires and encompass detailed mapping of fire severity patterns, the development of comprehensive emission factors, better models for growth and recovery across various vegetation types, an improved understanding of how fires of varying severities impact carbon stocks, and more in-depth analyses of human factors contributing to emissions, including ignition types and fire-weather conditions (Bowman *et al.*, 2023).

Another area of research examines the effectiveness of legislative proposals aimed at regulating GHG emissions following LULUCF. Within the EU-28, it was observed that there is a limited connection between the forest policy framework and its impact on land and forest owners (Nabuurs *et al.*, 2017). This sets forth a diverse range of opportunities to create positive incentives for more robustly integrate climate objectives into the forest sector. Furthermore, it highlights the potential of aligning forest climate goals with other forest-related concerns, such as biodiversity preservation, wildfire management, protection of peat carbon, addressing forest health issues, and low demand for raw wood materials (Nabuurs *et al.*, 2017).

Studies in China, the largest contributor to GHG emissions, have assessed the projections outlined in this country's INDC. When it comes to forests, research indicates that the goal set in the Chinese INDC to increase the forest stock volume by 4.5 billion cubic meters leads to a relatively modest reduction in carbon dioxide emissions compared to the emissions projected under current policy scenarios. This reduction is primarily attributed to the carbon accumulation in newly afforested areas and the carbon storage in existing forests (den Elzen *et al.*, 2016). It is important to note that there is a high level of uncertainty associated with these estimates, primarily related to how future harvest levels are projected and how the target is measured. These studies underscore the notion that achieving meaningful carbon reductions in China by or before 2030 would necessitate significant efforts to enhance GHG mitigation (den Elzen *et al.*, 2016).

4 Agriculture's Impact on Biodiversity and Wild/Nondomesticated Species

Biodiversity encompasses the entire spectrum of life, ranging from the molecular and organismic levels to populations, species, and entire ecosystems. In this way, it encompasses the full range of variations in the physical characteristics of individuals and populations within a species, the taxonomic variety of species within a community, the functional differences among species within an ecosystem, and the overall diversity found within the ecosystem itself (Hanley and Perrings, 2019). The surge in the global human population has resulted in approximately 40% of the Earth's forests and other ice-free natural habitats being transformed into farmland and grazing areas (Rey Benayas and Bullock, 2015). In fact, the primary driver of biodiversity loss is widely attributed to the expansion and intensification of agriculture (Barros-Rodríguez *et al.*, 2021; Bellard *et al.*, 2014; Kehoe *et al.*, 2017; Rasche *et al.*, 2022). As a result, scientists agree that the current era is witnessing a higher rate of species extinction when compared to previous eras in the planet's history (Rey Benayas and Bullock, 2015).

Agriculture's impact on biodiversity is classified as (i) direct associated with the alterations in soil composition, land-use-change, use of pesticides, etc., and (ii) indirect, such as the evolutionary impacts on domesticated species (Turcotte *et al.*, 2017). Regarding the mechanisms through which agriculture influences wildlife evolution, three distinct mechanisms have been identified (Turcotte *et al.*, 2017). The first, involves species adapting to a different set of genetic traits, as domesticated species often exhibit vastly different phenotypes compared to their wild ancestors. For instance, combatting pests in agriculture often involves breeding domesticated crop varieties with enhanced disease resistance. However, pests can quickly adapt to these new resistant varieties. The second mechanism is that intensive agricultural practices, such as crop rotation, pesticide use, and genetic engineering, exert selective pressure on the evolution of wild species (Turcotte *et al.*, 2017). The high density of domesticated species, along with irrigation, fertilization, and tillage, can result in agricultural habitats lacking biodiversity. This homogeneity can lead to rapid pest evolution. Efforts to control pests, weeds, and diseases also drive significant selection of wild organisms. The third mechanism involves non-selective processes that influence the evolution of wild species. One example is the genetic exchange between domesticated species and their wild relatives, which can lead to “genetic homogenization.” This gene flow between domestic and wild-related species is a subject of concern for some scientists (Turcotte *et al.*, 2017). To this latter point some sectors of the scientific community consider genomics as one reasonable pathway toward mitigating agriculture’s environmental impact. Yet the concerns raised in this paragraph should be considered to mitigate any potential impact of using genomics on species biodiversity.

Research examining biodiversity changes has identified several key factors that will have a major impact on terrestrial ecosystems. These factors include land-use changes, climate change, nitrogen deposition, biotic exchange, and elevated carbon dioxide concentrations. Notably, ecosystems with Mediterranean climates and grassland characteristics are expected to undergo more substantial transformations when compared to northern temperate ecosystems (Sala *et al.*, 2000). Other studies have projected that the combined effects of land use and climate change are anticipated to result in a 10% loss of habitat for large mammals, particularly in regions like Europe. While shifts in human consumption patterns are predicted to have a positive impact on habitat conservation efforts, these changes may not be sufficient to mitigate the overall risk of extinction for these species (Rondinini and Visconti, 2015).

At present, society faces a dual challenge: the need to accommodate a growing population with an apparent need of increasing food production, while also addressing the decline in biodiversity and ecosystem service degradation (Tscharntke *et al.*, 2012). However, scientists also argue that the issue of food security is more related to food distribution and efficient use, more than increasing the total amounts of food to be produced (U.N. Food and Agriculture Organization, 2021).

Some scientists argue that due to this necessity of providing food for the global population, proposing conservation-driven recommendations like restricting agricultural expansion or withdrawing land from production would be impractical (Martin *et al.*, 2020). Others claim that the foundation of worldwide food security relies on small-scale farmers rather than large commercial farms. This is due to the fact that approximately 80% of the world's hungry population resides in developing nations, with nearly 50% of them being smallholders, who cultivate land covering less than 2 hectares (Tscharntke *et al.*, 2012).

Agricultural expansion involves clearing and breaking up once-undisturbed ecosystems, resulting in edge habitats that lead to greater human encroachment on wildlife. This expansion leads to homogeneous landscapes, involving low plant diversity with higher alterations in biomass and soil microbial activity associated with improved crop production (Barros-Rodríguez *et al.*, 2021; Glidden *et al.*, 2021). The reduction of the availability of natural resources for wildlife is triggered by the introduction of pesticides, fertilizers, and antimicrobial compounds into the environment (Glidden *et al.*, 2021). Further, irrigation systems, which are extensively employed in agriculture within arid and semi-arid regions, have a significant impact on the nutritional composition of the soil. This can result in changes in the microbial community present. Specifically, irrigation systems that utilize wastewater treatment plants have been found to diminish the diversity of the soil microbiota. This reduction is accompanied by an increase in the levels of heavy metals and mercury within the soil, as well as the introduction of microorganisms originating from human sources (Barros-Rodríguez *et al.*, 2021).

In addition, agriculture exerts a direct impact on global biodiversity through practices like forest burning for crop cultivation and pasture establishment, which have profound effects on ecosystems. This practice, often exacerbated by wildfires, results in significant alterations to the chemical and physical properties of the soil. One notable change is the chemical oxidation of soil organic matter, which in turn affects the composition of soil microorganisms, including those beneficial for plant growth such as rhizobacteria. Additionally, fires contribute to the increased soil erosion and nutrient loss (Barros-Rodríguez *et al.*, 2021).

Tilling and plowing are the standard agricultural practices used to enhance soil aeration and facilitate the mixing of fertilizers. However, these practices have significant impacts on both the biological and chemical properties of the soil (Barros-Rodríguez *et al.*, 2021). They can lead to changes in the soil's biological aspects, including shifts in the populations of organisms like earthworms and alterations in microbial biodiversity. These practices promote the decline of nitrogen-sensitive plant species, such as legumes and nonvascular plants, in the ecosystem (Barros-Rodríguez *et al.*, 2021).

Changes in local climatic conditions resulting from land use and land-use changes, particularly in agriculture, have a detrimental impact on the availability of suitable microclimates for insect diversity. The effects of land use and climate change on insect biodiversity exhibit spatial variability, with tropical species being more susceptible compared to their temperate counterparts (Outhwaite *et al.*, 2022). This discrepancy can be attributed to the fact that tropical regions have historically maintained relatively stable temperatures. Consequently, tropical species have experienced a narrower range of past climatic conditions and tend to exhibit narrower thermal niches when compared to temperate species (Outhwaite *et al.*, 2022). Research indicates that biodiversity loss is particularly pronounced in tropical regions such as the Amazon basin and Sub-Saharan Africa (Barros-Rodríguez *et al.*, 2021; Kehoe *et al.*, 2017). Kehoe *et al.* (2017) explain that the greatest risk of species loss due to agricultural land use is concentrated in less economically developed countries with rich biodiversity, where governmental conservation efforts are minimal. Conversely, in countries like the United States, which allocate substantial funds to biodiversity conservation, there has been a decrease in voluntary participation in programs such as the Conservation Reserve Program. This decline is attributed to the fact that these programs often face competition from subsidized crops like corn and soybeans (Cunningham, 2022).

The apparent discord between various forms of agriculture, particularly irrigated agriculture, and the goals of biodiversity and wildlife conservation, has reached critical junctures both locally and globally. The increasing use of land for agriculture, aquaculture, forestry, mining, transport, industry, commerce, and housing implies reductions in the wildlands, leading to a loss of habitats for wild species (Hanley and Perrings, 2019). The conversion of natural habitats to croplands and pastures has triggered a decline of 20–50% of the population of vertebrates, invertebrates, and plant species (Turcotte *et al.*, 2017). Other effects of agriculture on biodiversity are the collapse of pollinators due to pesticide use and agriculture-induced habitat loss (Turcotte *et al.*, 2017).

One well-documented consequence is the impact on wetlands (Lemly *et al.*, 2000). These wetlands hold significant value as wildlife habitats, serving as sanctuaries for resident wildlife and as crucial stopover for wintering/breeding sites for species like waterfowls and birds. These losses have adverse effects on local wildlife and send ripples through migratory species such as birds, with far-reaching global implications (Lemly *et al.*, 2000).

When examining strategies to reduce the impact of agriculture on biodiversity and wildlife, scientists present two distinct viewpoints. The first involves "land sparing" which advocates for the separation of land into either areas for nature preservation, and areas for intense agricultural production. The second is "land sharing" which promotes the integration of food production and conservation on the same land, often referred to as wildlife-friendly farming (Tscharntke *et al.*, 2012). This ongoing debate encompasses challenges. First, it challenges the notion that higher yields and biodiversity are inherently at odds when farms are effectively managed. Second, it questions the assumption that increased yield automatically leads to the preservation of land for nature. Third, it highlights the disruptive impact of conventional intensification on the beneficial functions of biodiversity and the overall environmental quality. The debate also must be aligned with the consensus that food security hinges more on efficiently distributing food rather than solely increasing food production. Increasingly, scientists are advocating for the development of efficient food distribution systems before considering high-input agricultural intensification (Tscharntke *et al.*, 2012).

Some scientists have brought up the concern that the conservation of biodiversity and wildlife should not be perceived as contradictory to human interests. An emerging area of research focuses on how conserving biodiversity and wildlife can potentially decrease the likelihood of zoonotic pathogens spilling over from wild animals leading to epidemics and pandemics in both humans and livestock (Glidden *et al.*, 2021).

4.1 The Economic Value of Biodiversity

Understanding the economic consequences of biodiversity loss is crucial for raising awareness and establishing a connection between the natural world and human well-being (Paul *et al.*, 2020). Gaining insight into these values would enhance decision-making processes concerning biodiversity. However, there is a shortage of studies that examine the economic worth of biodiversity, and it is imperative to establish a consensus regarding the appropriate approach or methodology to employ, given the intricate and context-specific nature of these relationships. When modeling the economic value of biodiversity, practitioners primarily focus on the complex interactions between biodiversity and ecosystem functionality, as well as the interrelationship between ecosystem functionality and the provision of ecosystem services (Hanley and Perrings, 2019).

Some researchers argue that integrating biodiversity into the ecosystem service concept presents challenges because biodiversity fulfills at least three distinct roles: it serves as a critical factor influencing ecosystem functioning, acts as an ecosystem in its own right, and directly impacts human well-being (Mace *et al.*, 2012). Paul *et al.* (2020) explain that the connection between biodiversity and economic worth exhibits variability and can assume diverse functional shapes, primarily featuring positivity and concavity. However, it can also be graphically represented by strictly either concave or convex curves. These intricacies arise from a combination of factors including the specific type of ecosystem services involved, the trade-offs being considered, the influences of input variables, and the type of utility function employed to represent human preferences.

The study by Hanley and Perrings (2019) indicate that the economic impact of biodiversity can be either direct or indirect contingent upon how changes in biodiversity affect human well-being. The direct values of biodiversity are typically assessed in terms of both use and non-use values. Use values are exemplified by the benefits derived from activities like recreational hunting of deer or birdwatching rare species in wetlands. Non-use values encompass the satisfaction that people derive from the knowledge of the existence of certain species, even if they never encounter them in the wild, such as the snow leopard or killer whales. Furthermore, when delving into the realm of indirect values associated with biodiversity, Hanley and Perrings (2019) highlight that biodiversity serves as a fundamental input in various ecosystems that ultimately benefits society. These indirect values are exemplified by the role of biodiversity in supporting essential ecosystem functions, such as the contribution of pollinator bees and other beneficial insects to agriculture that predate on damaging pests; and ultimately benefiting human populations.

Taking a closer look at the economic implications of agriculture on biodiversity, it is important to recognize that the evolution of wild species in response to agricultural practices can have both direct and indirect effects on the provision of multiple ecosystem services (Turcotte *et al.*, 2017). One long-acknowledged consequence of this interaction is the rapid evolution of pests, pathogens, and weeds, which can lead to significant reductions in crop production. An illustrative example is the development of pesticide resistance among tobacco budworms in Texas and northern Mexico in 1970, which ultimately resulted in the abandonment of 285,000 hectares of cotton cultivation. This problem is not limited to a single region, globally crop losses attributable to pests, pathogens, and weed amount to a substantial 25–40% of the production of most crucial food crops. These damages have the potential to erode grower profits and have a substantial impact on both global food supply and human nutrition (Turcotte *et al.*, 2017). To combat the evolution of resistance among crop pests, growers are often compelled to apply greater quantities of pesticides. However, this response can have indirect consequences on numerous ecosystem services, further complicating the intricate relationship between agriculture, biodiversity, and ecosystem functions (Turcotte *et al.*, 2017).

5 Agriculture's Impact on Water Quality and Quantity

The UNESCO World Water Assessment Programme (2019) indicated that by 2050, water demand is expected to increase by 20–30% above its current levels. As of 2019, over 2 billion people experienced high water stress, and over 4 billion experienced severe water scarcity at least once a year. This trend is expected to continue as climate change intensifies. The variation in water availability across different locations and times is influenced by intricate natural factors such as hydrology, water demand, geographical characteristics of river systems, as well as by human factors like population growth, water resource development, and river management (Steinfeld *et al.*, 2020). These factors are in a constant state of flux and can change rapidly, impacting livelihoods, industries, and freshwater ecosystems that rely on water. In many situations, hydrological patterns do not remain constant over time, even though global water resource management strategies often make this assumption (Steinfeld *et al.*, 2020).

Rosegrant *et al.* (2009) reported that the irrigation water supply reliability (IWSR) index is expected to decline from 0.71 globally in 2000 to 0.66 by 2050. The stressors fueling the decline in IWSR include the expected increased competition from non-irrigation water demands, mainly from developing countries. Another stressor is the expansion of the livestock industry, considering the water used by animals and the water needed to grow crops to feed livestock. Climate change is expected to increase the magnitude of the problem, given recurrent volatility in rainfall levels and severe droughts.

Irrigated agriculture absorbs about 70% of all the planet's freshwater withdrawals, including rivers, lakes, and aquifers (Rosegrant *et al.*, 2009). Yet irrigation accounts for only 10% of the global agriculture water use, suggesting that most agriculture is still dependent on rain (Assouline *et al.*, 2015). Irrigation in agriculture is crucial for crops to achieve optimal yields. For example, along with other inputs, irrigated cereal crop yields are 60% higher than rainfed crops. With the expected global population increase, food security relies on irrigated crops; for example, 53% of cereal production growth during 2000–2050 will likely come from irrigated agriculture (Rosegrant *et al.*, 2009). Enhancing the climate resilience of irrigated agriculture is necessary, particularly in regions where the looming challenges of future water shortages or heightened water supply unpredictability are evident (Ward, 2022).

The issue concerning water encompasses not only its availability but also its quality. Water pollution includes salinization, microbiological contamination, eutrophication, excess nutrients, acidification, metal pollutants, toxic wastes, saltwater contamination, thermal pollution, and suspended solids. They also include natural contaminants such as arsenic and fluoride (Rosegrant *et al.*, 2009). Crop operations, livestock feeding, cropland, and pasture runoff, constitute the largest nitrogen source for freshwater and marine ecosystems (Stephenson and Shabman, 2017; Vilas *et al.*, 2020). This overabundance of nitrogen cause the degradation of the water quality and eutrophication of groundwater, rivers, lakes, and coastal and marine ecosystems, resulting in the loss of biodiversity and hypoxia (Mekonnen and Hoekstra, 2015). Mekonnen and Hoekstra (2015) estimated that the accumulated global gray water footprint (GWF) for 2002–2010 was $13 \times 10^{12}\,\mathrm{m}^2/\mathrm{yr}$. GWF is an indicator of human appropriation of freshwater resources. Researchers estimated that China absorbed 45% of the global total, and 75% of the GWF was from agriculture, 23% from domestic point sources, and 2% from industrial point sources. Out of agricultural activity, 18% of the GWF came from cereal cultivation, 15% from vegetables, and 11% from oil crops.

Irrigation systems with marginal water sources and inadequate drainage often lead to soil salinization (Assouline *et al.*, 2015; Rosegrant *et al.*, 2009). This problem is enhanced in arid regions with high population growth where irrigation expansion is highly needed or with practices incorporating more fertilizers than required in the irrigation water (Assouline *et al.*, 2015). About 60% of the groundwater worldwide withdrawn is used for agriculture (Lawell, 2016; Rosegrant *et al.*, 2009).

Efforts to mitigate agricultural risks aim to manage unexpected water-related losses and should be considered when formulating policies. These methods encompass a range of strategies, including substituting livestock forage, increasing dam storage capacity, improving reservoir management, implementing managed aquifer recharge, developing supplemental irrigation systems, adopting desalination techniques, promoting water conservation, establishing equitable water pricing, facilitating water trading, stream adjudications, and implementing carbon sequestration policy initiatives (Ward, 2022). The significance of approaches such as hydroeconomic analyses is growing in importance to evaluate the efficiency of these agricultural risk reduction methods (Ward, 2022).

6 Pesticides' Impact on the Environment

Pesticides for this manuscript encompass substances used as insecticides, fungicides, herbicides, rodenticides, molluscicides, and nematicides in agriculture. Pesticides are extensively used in modern agriculture, serving as a cost-effective and efficient approach to reduce losses, boosting agricultural productivity and improving the economic feasibility of farming (Bourguet and Guillemaud, 2016; Sharma *et al.*, 2019). The benefits of pesticides for humans rely on helping control insect vectors that could transmit deadly diseases. Also, in agriculture, they are employed to control plant weeds, insects, and diseases, increasing crop productivity. Scientists estimate that the absence of pesticide use would result in a loss of 78% of fruit production, 54% of vegetable production, and 32% of cereal production (Lamichhane, 2017).

While pesticides offer significant benefits for crop productivity, their widespread use can have serious consequences due to their potential for biomagnification and their persistent nature (Sharma *et al.*, 2019). The excessive utilization of pesticides and the associated pollution lead to the destruction of biodiversity, threatening birds, aquatic organisms, and animals (Kaur *et al.*, 2019; Mahmood *et al.*, 2016). Scientists claim that out of the three billion kilograms of pesticides used worldwide yearly, only a small percentage is used to control plant pests and the remaining is lost to non-target plants and the environment (Tudi *et al.*, 2021). Pesticides contaminate the air, water, soil, and entire ecosystems resulting in significant health risks for all living organisms (Sharma *et al.*, 2019). Worldwide, China is the major user of pesticide, followed by the United States and Argentina (Sharma *et al.*, 2019).

When pesticides are applied to a target plant or when they are disposed of, they could enter the environment causing harm. When pesticides enter the environment, they transfer and degrade, generating new chemicals. The transfer process to non-target plants and the environment happens via adsorption, leaching, volatilization, spray drift, and runoff (Tudi *et al.*, 2021). In China, a report suggests that 70% of the pesticides used are not taken up by plants but rather leach into the soil and groundwater (Zhang, 2018).

Pesticides can either be naturally occurring compounds or synthetically manufactured. Based on the chemical composition, pesticides are classified into groups namely organochlorine, carbamates, organophosphates, pyrethroids, and neonicotinoids (Kaur *et al.*, 2019). Pesticide exposure can have sublethal effects on terrestrial plants and can be lethal to non-target plants. When herbicides drift or volatilize, they can harm nearby trees and shrubs, making plants more susceptible to diseases and reducing seed quality (Mahmood *et al.*, 2016). Additionally, the use of broad-spectrum insecticides like carbamates, organophosphates, and pyrethroids can lead to significant declines in populations of beneficial insects such as bees and beetles (Mahmood *et al.*, 2016; Tudi *et al.*, 2021). There is suspicion that pesticide use contributes to declines in bird populations, as pesticides can accumulate in the tissues of bird species, leading to their death. Furthermore, fungicides can indirectly reduce bird and mammal populations by killing earthworms, which serve as a food source for these animals (Mahmood *et al.*, 2016).

One significant issue associated with pesticide overuse is the leaching, diffusion, volatilization, erosion, and run-off of these chemicals into the soil, which can adversely affect the microorganisms inhabiting it (Pérez-Lucas *et al.*, 2019). The leaching rate of pesticides is influenced by a range of factors, including the chemical properties of the pesticide, soil permeability, soil texture, organic matter content, volatilization, crop root uptake, and the method and dosage of pesticide application. These diverse factors collectively determine the extent to which pesticides may move through the soil and potentially impact the environment (Pérez-Lucas *et al.*, 2019). Soil-dwelling microbes play crucial roles in supporting plant health, including facilitating nutrient absorption, breaking down organic matter, and enhancing soil fertility (Mahmood *et al.*, 2016). Many soil microbes are involved in processes like converting atmospheric nitrogen into nitrates. Chlorothalonil and dinitrophenyl fungicides, for instance, have been found to disrupt the activities of nitrification and denitrification bacteria. Herbicides like glyphosate can inhibit the growth and activity of nitrogen-fixing bacteria in the soil. Moreover, herbicides can harm fungal species such as mycorrhizal fungi, which are essential for nutrient uptake by plants. Pesticides and fungicides also exhibit neurotoxic effects on earthworms, which are vital contributors to soil fertility (Mahmood *et al.*, 2016).

Water contaminated with pesticides poses a significant threat to aquatic ecosystems and the various life forms they support (Mahmood *et al.*, 2016). Pesticides pollute surface waters (e.g., rivers, lakes, streams, reservoirs, and estuaries) by directly applying pesticides to control aquatic weeds and insects (Mahmood *et al.*, 2016; Tudi *et al.*, 2021). Another pathway is the percolation and runoff, drifts, wastewater, and wastewater from clean-up equipment used for pesticide formulation and application (Mahmood *et al.*, 2016; Tudi *et al.*, 2021). Pesticides present in water can have detrimental effects on aquatic plants by reducing the dissolved oxygen levels, leading to physiological and behavioral alterations in fish populations. The use of herbicides can lead to eliminate aquatic plants resulting in sharp declines in oxygen levels, suffocating fish, and decreasing their productivity (Mahmood *et al.*, 2016). The harmful impacts are not limited to fish alone; other species like amphibians are also significantly affected by pesticides in contaminated surface waters, exacerbating the challenges they face due to overexploitation and habitat loss (Mahmood *et al.*, 2016).

Pesticides have harmful effects on human health, which can occur through both direct occupational exposure and indirect exposure. Farmworkers in fields and greenhouses, as well as those working in the pesticide industry, are at risk of occupational exposure (Bourguet and Guillemaud, 2016). Additionally, the families of farmers and individuals living in rural areas where pesticides are heavily used are indirectly exposed. The general population can also be indirectly exposed through the consumption of food and drinking water contaminated with pesticide residues. The health effects on humans include acute toxicity, carcinogenicity, reproductive and neurodevelopmental disorders, and disruption of the endocrine system (Bourguet and Guillemaud, 2016).

Climate change is altering the occurrence of pests in crops. Increased atmospheric carbon dioxide concentrations likely alter the plant's insect susceptibility, affecting pesticide applications. Also, increased carbon dioxide concentrations and changes in the nitrogen content in the soil will probably change insect distribution, and densities, altering the way pesticides are applied. Moreover, changes in precipitation patterns, likely wetter conditions, will increase the severity of insect infestations (Tudi *et al.*, 2021).

The primary determinant of pesticide usage is the presence and severity of weeds, pests, and diseases in a crop. These organisms are influenced by climate change, which can lead to genetic adaptation, changes in phenology, or shifts in geographical distribution (Delcour *et al.*, 2015). As temperatures rise, there is a growing likelihood that larger quantities of pesticides will be applied, with greater intensity in terms of increased amounts, doses, frequencies, and the use of different varieties or types of pesticide products to manage these evolving challenges (Delcour *et al.*, 2015).

Innovations incorporating integrated pest management principles are needed to mitigate pesticide pollution. Examples of these technologies include the application of insecticides not harmful to beneficial insects, biological control, varietal innovations (varieties of crops with decreased susceptibility to pests and diseases), precision agriculture, and diagnostic tools based on molecular methods (Lamichhane, 2017). Scientists believe that successful pollution mitigation involves a comprehensive system approach with a deep understanding of the complex dynamics of pests, plants, beneficial insects, and agronomic and cultural practices (Lamichhane, 2017).

Reducing pesticide use is a pivotal factor in preserving both the environment and human health. The adoption of new production strategies can effectively lead to a decrease in pesticide usage. Research has shown that low pesticide use, seldom results in decreased productivity and profitability in arable farms (Lechenet *et al.*, 2017). A study analyzed apple and pear farmers' willingness to adopt environmentally friendly pesticides, and found that this decision is often affected by how farmers' actions are perceived by others (Gallardo and Wang, 2013). Additional studies have found that farmers often believe they have limited capacity or autonomy to reduce pesticide usage and that successful examples, whether through peer farmers or access to knowledge, are necessary to bring about behavioral changes (Bakker *et al.*, 2021).

7 Selected Approaches to Assess Agriculture's Environmental Impact

Gaining a precise understanding of agriculture's effects on the environment is crucial. It is only through this understanding that effective adaptation and mitigation strategies can be developed. Assessing agriculture's environmental impact is a complex task that involves the evaluation of various factors and often employs different modeling and methodologies. The choice of model depends on the specific research question, data availability, and the scale of the assessment (e.g., farm-level, regional or global). Often a combination of models and approaches is used to provide a comprehensive understanding of agriculture's environmental impact. In this section, we examine nine different approaches to assess environmental impact from agricultural activities: life cycle assessment, environmental impact assessment, carbon footprint models, soil health (SH) models, water quality models, biodiversity modeling, economic and sustainability models, remote sensing and geographic information systems, and land-use change models.

7.1 Life Cycle Assessment (LCA)

LCA encompasses all environmental burdens associated with the processes producing a product or service, from the very origins of the raw materials and extending all the way to waste disposal (Klöpffer, 1997). LCA applications to agriculture are not the exception. The assessment must include the entire agricultural production system: including all the impacts from on-field activities and those related to the production of raw materials such as minerals and fossil fuels and farm inputs like fertilizers, plant protection substances, machinery or seeds (Brentrup *et al.*, 2004; Fan *et al.*, 2022; Haas *et al.*, 2000). These assessments should include manufacturing, transport, and distribution, until the end of the agri-food products end cycle (Perminova *et al.*, 2016).

The LCA has been standardized by ISO 14040 (ISO, 2006a) and 14044 (ISO, 2006b). Typically, the LCA encompasses a series of stages including the establishment of the system and its objectives, the compilation of the life cycle, the evaluation of the environmental impacts, a cross-cutting phase of interpretation, and the formulation of recommendations for improvement (Acosta-Alba *et al.*, 2019).

Multiple applications of LCA have been developed through years. LCA applications in wheat gran production determined that the primary environmental impact was the fertilizer application. This fertilizer application was identified as a significant driver of eutrophication, acidification, and the exacerbation of climate change. These studies also reveal that farm inputs production and transportation have comparatively smaller effects on the overall environmental footprint of wheat production (Brentrup *et al.*, 2004). Other LCA applications have yielded comprehensive maps on a continental to global scale, highlighting the essential aspects of agricultural production, with particular focus on fertilizer inputs (Ciais *et al.*, 2010; Potter *et al.*, 2010). Contemporary national-scale life cycle inventories exhibit a growing inclusion of regional-level data, enhancing the accuracy and granularity of LCA (Colomb *et al.*, 2015; Eady *et al.*, 2014).

Research also underscores the significance of integrating spatial analysis into LCA. Models such as SEAMLESS, for instance, encompass a comprehensive analysis of 12 European grown crops at the scale of EU-15, with improved spatial resolution (van Ittersum *et al.*, 2008). Additional research enhances spatial assessments of LCA, by incorporating farm books with detailed input utilization data and utilizing remote sensing-derived land-use maps in inventory analyses. These studies contribute to a more finely detailed spatial evaluation of agriculture's contributions to GHG, both at the national and continental scales (Navarro *et al.*, 2016). Approaches that complement LCA such as the LCA for Climate Smart Agriculture, aim to emphasize the advantages that technical options offer to agricultural production systems. This research highlights the need to advocate for climate-smart agriculture along with multicriteria environmental assessments (Acosta-Alba *et al.*, 2019).

7.2 Environmental Impact Assessment

The Environmental Impact Assessment (EIA) is a systematic process used to assess the environmental effects of a project, plan, or policy before its implementation. When applied to agriculture, EIA assesses the environmental risks associated with agricultural practices, encompassing crop production and livestock. The EIA rests on the principle that the impact of a human activity is contingent upon the pollution emanating from the activity and the environments' susceptibility to these actions (Payraudeau and van der Werf, 2005). EIA typically covers a wide range of environmental goals concerning agricultural inputs, including the effects of fertilizers and pesticides, as well as addressing GHG emissions. Also encompasses the overall effects of agriculture on ecological systems, including biodiversity, air quality, SH, and water quality (Kross *et al.*, 2022).

Important to note is that the boundaries of EIA are not rigidly defined. While some researchers employed it in conjunction with other assessment methods, others incorporate these alternative methods into the EIA framework. Some supplementary methods include the environmental risk mapping, LCA, multiagent system, linear programming, and the utilization of agro-environmental indexes (Payraudeau and van der Werf, 2005). The environmental risk assessment serves as a complementary process to EIA. While EIA is forward-looking, comparative, and concerned with evaluating all potential environmental effects, including secondary and tertiary indirect consequences, environmental risk assessment focuses on evaluating the likelihood of a specific adverse outcome arising from human activities (Jørgensen *et al.*, 2005). The EIA indexes encompass agricultural activities effects on climate change, ozone depletion, acidification, freshwater and marine eutrophication, toxicity effects on humans, terrestrial, freshwater and marine eco-toxicity, land use and fossil energy consumption, among others (Fan *et al.*, 2022).

An EIA application in Brazil sought to assess the impact of technology innovations at the farm level. The approach involved the creation of a series of weighting matrices, and field assessments that were carried out through interviews and surveys to farmers. These assessments were assigned weights according to the spatial scale and significance in influencing environmental impacts. The evaluation of these indicators formed an Environmental Impact Index, which served as a guiding framework for the implementation of agricultural technology innovations (Rodrigues *et al.*, 2003). A study analyzed the EIA application to the evaluation of Bangladesh's water sector. It was found that the standardization of EIA guidelines considering the country's social, economic, and political context, are necessary to formulate effective recommendations (Momtaz, 2002). Also, EIA was employed to analyze soybean farming practices in Argentina. It revealed that the timing of planting and fertilizer application played a crucial role in the leaching losses into groundwater. When considering no-tillage practices for soybeans and other crops, no evidence of soil degradation was found in soybeans. It was observed that crop rotations could potentially elevate the risk of nitrogen leaching compared to monoculture soybean. And that to monitor the environmental consequences of crop rotation, it was recommended a combination of remote sensing and modeling techniques (Kroes *et al.*, 2017).

7.3 Carbon Footprint Models

"Carbon footprint" is a significant metric for measuring GHG intensity associated with various activities and products, especially agriculture. Standard methodologies for calculating carbon footprints have been formulated and sector-specific guidelines are being developed (Pandey and Agrawal, 2014). Research shows that many of the existing calculators to measure GHG emission of agricultural products exhibit high levels of uncertainty and potentially could fail in detecting mitigation options along the production chain. To avoid this, scientists agree that calculators should consider the differences in pedoclimatic conditions, agricultural management practices and characteristics or perennial crops and crop rotations (Peter *et al.*, 2017). For instance, not accounting for the impact of crop rotation and the inclusion of crop residues as co-products in LCA and carbon footprint assessments could result in gross underestimations of GHG savings worldwide (Brankatschk and Finkbeiner, 2017).

Applications of carbon footprint assessments combined with other methodologies such as the Data Envelopment Analysis demonstrate that this combined approach functions as a practical tool to improve efficiencies of agricultural operations while reducing environmental impacts (Rebolledo-Leiva *et al.*, 2017). Carbon footprint and carbon footprint intensity measuring reveal that factors such as GDP per capita, planting structure, population density, and urbanization levels influence the spatiotemporal and intraregional heterogeneity of GHG agricultural emissions (Cui *et al.*, 2022).

7.4 Soil Health Models

Soil health (SH) involves the understanding that soil is the growing medium for crops and the foundation for essential ecosystem services.

Measuring SH requires the development of holistic indexes or assessment frameworks to evaluate the effects of soil management practices and land uses (Rinot *et al.*, 2019). However, scientists agree that developing frameworks is complex because of the site-specificity of terrestrial ecosystems, and the many linkages between soil functions and soil-based ecosystem services (Bünemann *et al.*, 2018).

There is an abundance of methods relying on analytical approaches across different countries. The national soil quality monitoring in Canada to assess the status and trends in SH, and to assess inherent soil quality and susceptibility change (Acton and Gregorich, 1995; MacDonald *et al.*, 1998; Wang *et al.*, 1997). In the United States, two quality assessments were developed: the soil management assessment framework whose objective was to evaluate management practices (Andrews *et al.*, 2004; Karlen *et al.*, 2001; Wienhold *et al.*, 2004, 2009). Also, the Cornell test to assess SH, address soil degradation, and increase productivity (Idowu *et al.*, 2008; Moebius-Clune, 2016). In Australia, researchers developed the soil quality website mainly to establish target, threshold values for benchmark sites (Gonzalez-Quiñones *et al.*, 2015). In New Zealand, the 500 soils project analyzes soil quality for environmental reporting (Lilburne *et al.*, 2004; Schipper and Sparling, 2000; Sparling *et al.*, 2004; Sparling and Schipper, 2004). In France, scientists developed a framework to assess soil quality for environmental protection, food security, and sustainable management practices (Antoni *et al.*, 2007; Arrouays *et al.*, 2003, 2002; Martin *et al.*, 1998). In the United Kingdom, an approach to assess soil functions of environmental interactions were developed (Loveland and Thompson, 2002; Merrington *et al.*, 2006). In Ireland, the soil quality assessment research project developed an assessment of soil functions (Bondi *et al.*, 2017). In the Netherlands, the National Soil Quality Monitoring Network developed an assessment of soil quality and land-use effects (Wattel-Koekkoek *et al.*, 2012). The European Soil Monitoring and Assessment Framework was formed to provide objective, reliable, and comparable information at the European level (Huber *et al.*, 2001).

The majority of these frameworks were originally crafted by soil and environmental scientists. Nevertheless, owing to their policy implications, economists advocate for the incorporation of economics into the formulation of these frameworks and indices. Economists can make valuable contributions by evaluating the cost-effectiveness and benefits of policies related to SH (Stevens, 2018).

7.5 Water Quality Models

Water quality models seek to discover methods to address the degradation or excessive use of water resources (Tsakiris and Alexakis, 2012). Also, these models play a crucial role in detecting water pollution and understanding the ultimate fate and behaviors of contaminants within aquatic systems (Wang *et al.*, 2013). Others argue that water quality models serve as tools for assessing the influence of land management, land use, climate, and conservation practices on water resources, ecology, and the provision of ecosystem services related to water (Moriasi *et al.*, 2015).

The following models are primarily employed to assess water quality. For a more detailed review of these models please see Tsakiris and Alexakis (2012). DRAINMOD is a well-known water evaluation model, which is designed to analyze the functioning of subirrigation and drainage systems and their impacts on pollutant removal, crop behavior, and water utilization. For a comprehensive list of all DRAINMOD-related publications, see the Agricultural Water Management from North Carolina State University (*Source*: https://www.bae.ncsu.edu/agricultural-water-management/drainmod/drainmod-publications/).

The Export Coefficient Model developed by the University of Reading has widely been used to predict the total amount of phosphorus and nitrogen delivered to any given surface water sampling site (Bowes *et al.*, 2008; European Commission, 2003; Ierodiaconou *et al.*, 2005; Johnes, 1996). MIKE-11 was developed by the Danish Hydraulic Institute and is a hydrodynamic model primarily utilized for simulating the one-dimensional flow of water in rivers, and it has seen widespread application in Northern India and England (Crabtree *et al.*, 1996; Kazmi and Hansen, 1997). The Modelling Nutrient Emissions in River Systems (MONERIS) was created by the Leibniz Institute for Freshwater Ecology and Inland Fisheries; and relies on runoff water quality data for the study area in conjunction with Geographic Information System (GIS) (Venohr *et al.*, 2011). The Simulation of Catchments (SIMCAT) (Warn, 2010) and the Temporal/Overall Model for Catchments (TOMCAT)

(Bowden and Brown, 1984; Cox, 2003) are models that use Monte Carlo analysis techniques and were developed by the U.K. Environmental Agency. The QUAL2K is a steady-state model of water quality and in-stream flow and was developed by the U.S. Environment Protection Agency (Brown and Barnwell, 1987).

A meta-analyses of performance measures and performance evaluation criteria of watershed-scale models is presented in Moriasi *et al.* (2015). Other analyses including performance evaluation of water quality models can be found in Harmel *et al.* (2014), Bennett *et al.* (2013), Biondi *et al.* (2012), Black *et al.* (2014), Pushpalatha *et al.* (2012), and Ritter and Muñoz-Carpena (2013).

7.6 Biodiversity Modeling

Biodiversity models are mathematical-based computations that simulate, analyze, and predict patterns and processes related to biodiversity in ecosystems (Chave and Thebaud, 1988).

The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) envisions a comprehensive classification of models based on both the nature of the relationships being modeled and the methods employed to model these relationships. When considering the relationships being modeled, there are three distinct categories of models: (i) those that examine the repercussions of alterations in indirect drivers, such as sociopolitical, economic, and technological factors; (ii) those that investigate the direct drivers of changes in nature, such as land-use transformations, climate variations, and nitrogen deposition; and (iii) models that assess the consequences of changes in biodiversity and ecosystems on human populations. Additionally, models can be categorized based on their methodology, including correlative models, process-based models, and expert-based models (Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services, 2019).

The models assessing biodiversity change address aspects such as species extinctions, species abundance, community structure, habitat deterioration, and alterations in the distribution of species and biomes (Pereira *et al.*, 2010). A subcategory are the species-area relationships models that primarily center on predicting the potential loss of species due to habitat alterations (Daily *et al.*, 2001; Pereira and Daily, 2006).

Another category of models includes distribution species models, that employ complex tools such as site-based ecology and spatial data technologies to analyze and simulate the geographic distribution of terrestrial, freshwater, and marine species across different spatial and temporal scales (Elith and Leathwick, 2009). More specifically, distribution species models are used to draw inferences about species' range boundaries and the suitability of habitats. Various approaches are employed, with many of them being correlative models that establish connections between spatial data and species distribution models (Kearney and Porter, 2009). Studies focus on enhancing existing models, with one key concern in species distribution models being their transferability — which refers to the model's capacity to accurately predict biodiversity in new environments (Sequeira *et al.*, 2018). Additional research focus on refining models that forecast both present and future pattern of species composition and richness. A topic of ongoing discussion in these studies revolve around how to account for the potential saturation of environments in biodiversity models (Mateo *et al.*, 2017).

Biodiversity models are critical to improve the understanding the vast variety of functional and taxonomic forms of life, the causes of its existence and how to preserve them (Chave and Thebaud, 1988). IPBES argues that the importance of biodiversity modeling stems on gaining insights on how biodiversity responds to environmental changes. Therefore, modeling is important to assess and predict the impacts of drivers on biodiversity and ecosystems, and its ultimate impact on human well-being (Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services, 2019). These models use data from field surveys, remote sensing, and ecological frameworks to estimate parameters and validate predictions.

7.7 Economic and Sustainability Models

Economic models are analytical tools that help assess the economic outcomes of the relationships between agricultural activities and the environment. Their primal significance lies in their ability to inform policy analyses through the evaluation of potential policy outcomes. For instance, González-Ramírez *et al.* (2012) examined a collection of studies focused on assessing the effectiveness of carbon offset policies in agriculture and found that, when formulating carbon contracts, various approaches were considered. These approaches encompass per-ton contracts, which involve measuring the opportunity costs of providing carbon credits, as well as output-based offset mechanisms, principal-agent contracts, and dynamic stochastic abatement cost models.

When evaluating payment for ecosystem services programs to prevent deforestation and encourage forest growth, Alix-Garcia and Wolff (2014) reviewed several studies that analyzed the demand, supply, and indirect consequences of various programs. These studies revealed that, in theory, cost-effective contracts should compensate for the opportunity cost of using land with the highest anticipated net benefits. A method for estimating opportunity costs could involve the use of experimental auctions. Stephenson and Shabman (2017) reviewed studies assessing the effectiveness of water quality trading programs, and determined they would not have a substantial impact on agricultural nonpoint sources. Nonetheless, it was acknowledged that the implementation of market-like incentive policies within the water quality trading programs could greatly enhance the opportunities to positively and significantly affect water quality. Segerson (2013) conducted an extensive review of economic literature and discovered that voluntary programs can be effective in mitigating environmental harm in certain contexts, but only if they are carefully designed. More specifically, such programs have the potential for effectiveness if there is an underlying regulatory framework that supports the voluntary approach.

Börner *et al.* (2020) conducted a comprehensive review of the payment for ecosystem services policies, integrated conservation and development projects, and the protection of indigenous lands. It was determined that there exists limited understanding regarding the cost-effectiveness of these policies. Another relevant finding is that in some instances development policies could contribute to deforestation, undercutting the effectiveness of conservations measures. Segerson (2022) reviewed the effectiveness of group incentive conservation policies. This scheme involves the imposition of penalties or the granting of rewards to an individual based on the overall performance of an entire group. The study revealed that, in theory, both proportional schemes (where rewards/penalties vary in relation to performance) and fixed payment schemes (where fixed rewards/penalties are applied when certain thresholds are achieved) offer incentives for promoting efficient behavior. However, the actual results would hinge on the specific design of the policy and the dynamics of internal group interactions, including factors like trust, social capital, and leadership.

On more general issues, economic models have been used to assess the effects of climate change on the world economy. Studies applying micro and macro-economic modeling mixed with simulations have analyzed the potential impact of increased temperatures on national and global incomes, concluding that warming could amplify the global inequality because hot countries are usually poorer and they would experience the largest growth reductions (Burke *et al.*, 2015; Dell *et al.*, 2012; Stern, 2013).

Economic models such as the computable general equilibrium models also have been applied to assess the value of ecosystem services and impacts of climate change in agriculture and its adaptation (Jones *et al.*, 2017). Also, computable general equilibrium models have been applied to analyze climate change effects on crop yields and trade patterns, finding a negative productivity effect of climate change, triggering more intensive management practices, area expansion, fostering international trade and reduced consumption with the distribution of the magnitude of the effects varying across countries (Nelson *et al.*, 2014).

Economic models also help identifying optimal land use between agricultural production and environmental conservation. For example, early efforts to integrate farmers' choice drivers along with established ecosystem models aimed to better explain land-use decision making (Bockstael *et al.*, 1995). Different economic approaches are employed to assess land-use values, for instance the input–output models incorporating the costs of agricultural land and how changes impact farms (Münier *et al.*, 2004).

7.8 Remote Sensing and Geographic Information Systems

Remote sensing and geographic information systems provide valuable data enabling for soil property mapping, crop species classification, identification of crop water stress, monitoring crop diseases and weed infestations, and the creation of crop yield maps. The data is instrumental in directing sustainable land management practices and promoting the responsible utilization of natural resources within agriculture (Khanal *et al.*, 2017).

Remote sensing has the capability to perform land-cover characterization, mapping, and monitoring at the local and global scale. This advanced technology captures high-resolution images that can be used to classify and map different land cover categories, including agricultural land. This allows for the identification of areas under cultivation, crop types, and changes in land-use over time (Giri, 2012).

Furthermore, remote sensing can be used for monitoring crop health, as it can detect changes in crop health and stress levels through the analysis of vegetation indices. These indices play a crucial role in evaluating crop conditions, identify diseases, and optimize irrigation and fertilization (Doraiswamy *et al.*, 2003). Remote sensing also offers important information for management nutrient and water stress management. It aids in the assessments of SH and the optimization of irrigation practices, for an efficient use of resources (Shanmugapriya *et al.*, 2019). Moreover, a comprehensive understanding of soil moisture and its spatial and temporal variations hold significant relevance for a wide range of meteorological, climatological, and hydrological applications. This plays an important role in improving our understanding of the relationships within water, energy, and carbon cycles, as well as in forecasting extreme climate events (Babaeian *et al.*, 2019).

GIS, in conjunction with various technologies like remote sensing, global positioning system, artificial intelligence, computational systems, and data analytics; all have played a central role in crop monitoring and the implementation of precise and targeted management practices aimed at enhancing crop productivity (Ghosh and Kumpatla, 2022; Pierce and Clay, 2007).

GIS can be employed to create environmental risk maps by facilitating the integration of diverse environmental data, including soil quality, water resources, and land cover. This integration enables the assessment of the environmental consequences of agricultural activities (Payraudeau and van der Werf, 2005). GIS can be used in watershed management by enhancing our understanding of the water flow, sediment transport, and movement of pollutants from agricultural regions into watersheds (Lyon, 2002). GIS is also used to assess biodiversity conservation efforts by enabling the mapping and monitoring of habitats, wildlife corridors, and regions of significant biodiversity importance. This aids in identifying areas where agricultural activities could potentially encroach sensitive ecosystems, thus facilitating more effective conservation planning (Salem, 2003).

7.9 Land-Use Change Models

Land-use change models help assess how changes in land use, particularly related to agriculture, can affect the environment. The most widely applied models are cited in this section. The CLUE (Conversion of Land Use and its Effects) model is used to simulate and replicate land-use changes in space and time, stemming from the interplay of biophysical and human influences (Veldkamp and Fresco, 1996). The latter includes the effects of agricultural intensification, deforestation, land abandonment, and urbanization (Verburg and Overmars, 2009). Another model is LEAM (Land Use Evolution and Impact Assessment Model) that complements with other tools (e.g., GIS) to develop a planning support system to better understand land-use changes after the spatial and dynamic interaction among economic, ecological, and social systems (Deal *et al.*, 2005).

InVEST (Integrated Valuation of Ecosystem Services and Trade-offs) is a suite of models built on the idea that ecosystems services are often not well-understood, inadequately monitored and face rapid degradation and depletion. InVEST models help reconciling environmental and economic objectives for a wide range of organizations and stakeholders (Tallis *et al.*, 2010). DPSIR (Drivers-Pressures-State-Impact-Response) is a transdisciplinary tool used to analyze the environment by considering a cause–effect relationship among the interconnected components of social, economic, and environmental systems (Khajuria and Ravindranath, 2012). LUCI (Land Utilization Capability Indicator) is a specialized modeling tool that provides detailed spatial information to evaluate how alterations in land use can affect a wide range of ecosystem services. One of the model's primary objective is to enhance the understanding of city and landscape planners regarding the potential outcomes of different land use changes or interventions (Veerkamp *et al.*, 2023).

LCM (Land Change Modeler) is a land change projection tool used for land planning. This model utilizes historical land cover change data to create empirical models that depict how different land cover transitions are related to various explanatory variables. By doing so, it can generate maps illustrating potential future scenarios of land-use change (Camacho Olmedo *et al.*, 2018). SLEUTH (Slope, Land-use, Exclusion, Urbanization, Transportation, and Hillshading) operates as a cellular automaton model. A cellular automaton consists of individual cells organized in a grid of a specific shape. Each cell's state changes over time based on a predefined set of rules influenced by the states of neighboring cells. SLEUTH simulates urban growth over time and can also model changes across a range of land use categories (Camacho Olmedo *et al.*, 2018).

Insights from this section

The section discusses various approaches to assess the environmental impact of agricultural activities, showcasing applications and contributions. Evaluating impacts is a complex task. It is important to incorporate economic considerations into the different frameworks so it facilitates the integration into policy assessments. These approaches are also crucial for guiding agriculture adaptation and the implementation of sustainable practices.

8 Agriculture Adaptation

To complement the review presented in the article, it is key to emphasize how environmental shifts impact agricultural activities and the production of sufficient, high-quality food for an expanding population. Consistent climate patterns, including temperature, humidity, and rainfall are vital for optimal crop cultivation and livestock farming. In fact, raising temperatures have led to increased rates of crop respiration and evapotranspiration, as well as heightened occurrences of diseases, pest infestations, and weed proliferation (Carter *et al.*, 2018; Malhi *et al.*, 2021; Yasmeen *et al.*, 2022b). Thus, it is imperative to invest in adaptation practices to ensure the environmental sustainability, the economic sustainability of farm operations and to secure a steady food supply for an expanding global population (IPCC, 2014). The successful implementation of adaptation practices to improve food systems relies heavily on the formulation of effective policies. However, effective policy-making hinges on the availability of high-quality information, as knowledge gaps frequently hinder the successful implementation of policies. These gaps in knowledge can be in the form of disparities between national and regional scales, differences in the data availability at the farm level, varying frequencies of data collection ranging from yearly to monthly and the absence of cross-country harmonization, among other gaps (Deconinck *et al.*, 2021).

Adaptation involves modifying current practices in response to climate effects, aiming to mitigate the negative impacts and leverage any potential beneficial opportunities (Agrawala *et al.*, 2011). Doria *et al.* (2009) also emphasizes that adaptation must not undermine economic, social, or environmental sustainability. As such, successful adaptation strategies require investments in knowledge, meticulous planning, effective coordination, and a proactive approach (Fankhauser, 2017). Note that, the scope of adaptation is influenced by the economic environment, including market conditions and policy incentives that facilitate increased research and development, knowledge transfer, institutional support, and incentives promoting the efficient utilization of resources (Agrawala *et al.*, 2011).

It is important to note that adaptation and mitigation strategies can often intersect. From a policy standpoint, both are crucial, each serving a unique purpose. However, from an economic perspective, they are often viewed as substitutes because heightened adaptation efforts can potentially diminish the incremental benefits derived from mitigation practices (Fankhauser, 2017). In line with the economic perspective, it is essential to establish a connection between the concepts of adaptation and adoption at the micro level. Zilberman *et al.* (2012) suggests that adaptation can occur at the individual farm level through the adoption of technologies or by making changes in the use of inputs, in order to mitigate agriculture's impact on the environment. As such, adaptation in agriculture manifests in three primary forms: (i) adjustments in management practices, such as alterations in planting schedules, (ii) modifications in inputs, like the integration of more heat-tolerant crop varieties, and (iii) the incorporation of new technologies that demand capital investment and facilitate improved practices, such as the implementation of more efficient irrigation systems and enhanced fertilization techniques (McCarl, 2007).

Literature addressing agricultural adaptation primarily revolves around economic modeling, as it necessitates investment in novel technologies to optimize resource efficiency, as previously discussed. Various studies aimed at gauging the advantages of adapting agricultural practices to climate change have deduced that even modest levels of adaptation can counterbalance the downturn in agricultural output resulting from climate change (Adams *et al.*, 1995; Darwin, 1995; Kurukulasuriya and Mendelsohn, 2007; Reilly *et al.*, 1994; Rosenzweig and Parry, 1994; Seo and Mendelsohn, 2008; Tan and Shibasaki, 2003; Wang *et al.*, 2009).

9 Conclusions

Agriculture and the broader food production system play a pivotal role in sustaining the human species. However, our heavy reliance on land and water usage has led to the depletion of the environment and biodiversity. In this context, the question arises: can we produce sufficient food to meet the needs of a growing population while simultaneously mitigating the inevitable environmental impacts? Is it a matter of choosing between globalized productivism and industrial agriculture versus environmentally friendly farming? Are these concepts fundamentally incompatible?

Intensive agriculture and environmentally friendly farming should not be mutually exclusive. The concept of sustainable intensification, initially seemingly incompatible, appears to be a plausible approach. Both developed and developing countries, including all stakeholders in the agri-food chain, consumers, policymakers, academia, and society at large, must collaborate to integrate a healthy environment, economic viability, social and economic equity, and the capacity to adapt to climate change.

To foster sustainable agricultural practices, it is imperative to explore promising alternatives, such as systems-based approaches and the integration of technologies, alongside comprehensive policies. This necessity is particularly urgent in developing countries, given that the impacts of environmental disruptions are projected to be more pronounced in these regions. Despite the inherent economic motivation to adapt to climate change (as profits will inevitably decline without adaptation), the uptake and dissemination of mitigation technologies have been slower than anticipated. Experts concur that governmental interventions, through the establishment of incentive frameworks, policy instruments, and targeted programs, play a vital role in steering agriculture towards a more environmentally sustainable path. Within this context, ensuring political stability, implementing sound legal provisions, and effectively curbing corruption are crucial for the development and enforcement of effective measures (Yasmeen *et al.*, 2022a).

Ultimately, the existence of communication gaps between the scientific community and agricultural stakeholders hinders the comprehensive understanding of the full scope of climate change, thereby impeding the swift adoption of mitigation practices and technologies (Getson *et al.*, 2022). Encouraging climate scientists to undergo communication training and employ communication styles informed by social science research is recommended. Transparent communication becomes particularly effective when discussing the probabilities of potential scenarios, the spectrum of potential impacts, and the intrinsic uncertainty surrounding predictions.

References

Aan den Toorn, S., E. Worrell, and M. Van Den Broek. 2021. "How Much Can Combinations of Measures Reduce Methane and Nitrous Oxide Emissions from European Livestock Husbandry and Feed Cultivation?". *Journal of Cleaner Production*. 304: 127138.

Acosta-Alba, I., E. Chia, and N. Andrieu. 2019. "The lca4csa Framework: Using Life Cycle Assessment to Strengthen Environmental Sustainability Analysis of Climate Smart Agriculture Options at Farm and Crop System Levels". *Agricultural Systems*. 171: 155–70. https://doi.org/10.1016/j.agsy.2019.02.001.

Acton, D. F. and L. Gregorich. 1995. "The Health of Our Soils: Toward Sustainable Agriculture in Canada".

Adams, R. M., R. A. Fleming, C.-C. Chang, B. A. McCarl, and C. Rosenzweig. 1995. "A Reassessment of the Economic Effects of Global Climate Change on U.S. Agriculture". *Climatic Change*. 30(2): 147–67. https://doi.org/10.1007/BF01091839.

Agrawala, S., F. Bosello, C. Carraro, E. De Cian, and E. Lanzi. 2011. "Adapting to Climate Change: Costs, Benefits, and Modelling Approaches". *International Review of Environmental and Resource Economics*. 5: 245–84. https://doi.org/10.1561/101.00000043.

Aguirre-Villegas, H. A. and R. A. Larson. 2017. "Evaluating Greenhouse Gas Emissions from Dairy Manure Management Practices Using Survey Data and Lifecycle Tools". *Journal of Cleaner Production*. 143: 169–79. https://doi.org/10.1016/j.jclepro.2016.12.133.

Alix-Garcia, J. and H. Wolff. 2014. "Payment for Ecosystem Services from Forests". *Annual Review of Resource Economics*. 6: 361–80.

Alston, J. M. and P. G. Pardey. 2014. "Agriculture in the Global Economy". *Journal of Economic Perspectives*. 28(1): 121–46. https://doi.org/10.1257/jep.28.1.121.

Andrews, S. S., D. L. Karlen, and C. A. Cambardella. 2004. "The Soil Management Assessment Framework". *Soil Science Society of America Journal*. 68(6): 1945–62. https://doi.org/10.2136/sssaj2004.1945.

Antoni, V., N. Saby, C. Jolivet, B. Toutain, J. Thorette, and D. Arrouays. 2007. "The French Information System on Soils: A Decision Support System for Soil Inventory, Monitoring and Management". *EnviroInfo*. (1): 255–62.

Arrouays, D., C. Jolivet, L. Boulonne, G. Bodineau, C. Ratié, N. Saby, and E. Grolleau. 2003. "Le réseau de mesures de la qualité des sols (rmqs) de france". *Etude et gestion des Sols*. 10(4): 241–50.

Arrouays, D., C. Jolivet, L. Boulonne, G. Bodineau, N. Saby, and E. Grolleau. 2002. "Une initiative nouvelle en france: La mise en place d'un réseau institutionnel de mesure de la qualité des sols (rmqs)". *Comptes Rendus de l'Académie d'Agriculture de France*. 88(5): 93–103.

Assouline, S., D. Russo, A. Silber, and D. Or. 2015. "Balancing Water Scarcity and Quality for Sustainable Irrigated Agriculture". *Water Resources Research*. 51(5): 3419–36. https://doi.org/10.1002/2015WR017071.

Babaeian, E., M. Sadeghi, S. B. Jones, C. Montzka, H. Vereecken, and M. Tuller. 2019. "Ground, Proximal, and Satellite Remote Sensing of Soil Moisture". *Reviews of Geophysics*. 57(2): 530–616.

Bakker, L., J. Sok, W. Van Der Werf, and F. Bianchi. 2021. "Kicking the Habit: What Makes and Breaks Farmers' Intentions to Reduce Pesticide Use?". *Ecological Economics*. 180: 106868.

Banger, K., H. Tian, and C. Lu. 2012. "Do Nitrogen Fertilizers Stimulate or Inhibit Methane Emissions from Rice Fields?". *Glob Change Biology*. 18(10): 3259–67. https://doi.org/10.1111/j.1365-2486.2012.02762.x.

Barros-Rodríguez, A., P. Rangseekaew, K. Lasudee, W. Pathom-Aree, and M. Manzanera. 2021. "Impacts of Agriculture on the Environment and Soil Microbial Biodiversity". *Plants (Basel)*. 10(11). https://doi.org/10.3390/plants10112325.

Bashir, M. T., S. Ali, M. Ghauri, A. Adris, and R. Harun. 2013. "Impact of Excessive Nitrogen Fertilizers on the Environment and Associated Mitigation Strategies". *Asian Journal of Microbiology, Biotechnology Environmental Science*. 15(2): 213–21.

Bellard, C., C. Leclerc, B. Leroy, M. Bakkenes, S. Veloz, W. Thuiller, and F. Courchamp. 2014. "Vulnerability of Biodiversity Hotspots to Global Change". *Global Ecology and Biogeography*. 23(12): 1376–86. https://doi.org/10.1111/geb.12228.

Bennett, N. D., B. F. W. Croke, G. Guariso, J. H. A. Guillaume, S. H. Hamilton, A. J. Jakeman, S. Marsili-Libelli, L. T. H. Newham, J. P. Norton, C. Perrin, S. A. Pierce, B. Robson, R. Seppelt, A. A. Voinov, B. D. Fath, and V. Andreassian. 2013. "Characterising Performance of Environmental Models". *Environmental Modelling & Software*. 40: 1–20. https://doi.org/10.1016/j.envsoft.2012.09.011.

Biondi, D., G. Freni, V. Iacobellis, G. Mascaro, and A. Montanari. 2012. "Validation of Hydrological Models: Conceptual Basis, Methodological Approaches and A Proposal for A Code of Practice". *Physics and Chemistry of the Earth, Parts A/B/C*. 42–44: 70–6. https://doi.org/10.1016/j.pce.2011.07.037.

Black, D. C., P. J. Wallbrink, and P. W. Jordan. 2014. "Towards Best Practice Implementation and Application of Models for Analysis of Water Resources Management Scenarios". *Environmental Modelling & Software*. 52: 136–48. https://doi.org/10.1016/j.envsoft.2013.10.023.

Blujdea, V. N., R. A. Viñas, S. Federici, and G. Grassi. 2015. "The EU Greenhouse Gas Inventory for the LULUCF Sector: I. Overview and Comparative Analysis of Methods Used by EU Member States". *Carbon Management*. 6(5–6): 247–59.

Bockstael, N., R. Costanza, I. Strand, W. Boynton, K. Bell, and L. Wainger. 1995. "Ecological Economic Modeling and Valuation of Ecosystems". *Ecological Economics*. 14(2): 143–59.

Bondi, G., D. Wall, M. Bacher, J. Emmet-Booth, J. Graça, I. Marongiu, and R. Creamer. 2017. "Role of Soil Biology and Soil Functions in Relation to Land Use Intensity". In: *Egu General Assembly Conference Abstracts*. 15021.

Börner, J., D. Schulz, S. Wunder, and A. Pfaff. 2020. "The Effectiveness of Forest Conservation Policies and Programs". *Annual Review of Resource Economics*. 12(1): 45–64. 10.1146/annurev-resource-110119-025703.

Bourguet, D. and T. Guillemaud. 2016. "The Hidden and External Costs Of Pesticide Use". *Sustainable Agriculture Reviews: Volume*. 19: 35–120.

Bowden, K. and S. Brown. 1984. "Relating Effluent Control Parameters to River Quality Objectives Using A Generalised Catchment Simulation Model". *Water Science and Technology*. 16(5–7): 197–206.

Bowes, M. J., J. T. Smith, H. P. Jarvie, and C. Neal. 2008. "Modelling of Phosphorus Inputs to Rivers from Diffuse and Point Sources". *Science of The Total Environment*. 395(2): 125–38. https://doi.org/10.1016/j.scitotenv.2008.01.054.

Bowman, D. M., G. J. Williamson, M. Ndalila, S. H. Roxburgh, S. Suitor, and R. J. Keenan. 2023. "Wildfire National Carbon Accounting: How Natural and Anthropogenic Landscape Fires Emissions are Treated in the 2020 Australian Government Greenhouse Gas Accounts Report To The UNFCCC". *Carbon Balance and Management*. 18(1): 14.

Brankatschk, G. and M. Finkbeiner. 2017. "Crop Rotations and Crop Residues are Relevant Parameters for Agricultural Carbon Footprints". *Agronomy for Sustainable Development*. 37: 1–14.

Brentrup, F., J. Küsters, H. Kuhlmann, and J. Lammel. 2004. "Environmental Impact Assessment of Agricultural Production Systems Using the Life Cycle Assessment Methodology: I. Theoretical Concept of a LCA Method Tailored to Crop Production". *European Journal of Agronomy*. 20(3): 247–64.

Brown, L. C. and T. O. Barnwell. 1987. *The Enhanced Stream Water Quality Models Qual2e and Qual2e-Uncas: Documentation and User Manual*. US Environmental Protection Agency. Office of Research and Development.

Bryant, B. P., T. R. Kelsey, A. L. Vogl, S. A. Wolny, D. MacEwan, P. C. Selmants, T. Biswas, and H. S. Butterfield. 2020. "Shaping Land Use Change and Ecosystem Restoration in A Water-Stressed Agricultural Landscape to Achieve Multiple Benefits". *Frontiers in Sustainable Food Systems*: 4. https://doi.org/10.3389/fsufs.2020.00138.

Bünemann, E. K., G. Bongiorno, Z. Bai, R. E. Creamer, G. De Deyn, R. de Goede, L. Fleskens, V. Geissen, T. W. Kuyper, P. Mäder, M. Pulleman, W. Sukkel, J. W. van Groenigen, and L. Brussaard. 2018. "Soil Quality — A Critical Review". *Soil Biology and Biochemistry*. 120: 105–25. https://doi.org/10.1016/j.soilbio.2018.01.030.

Burke, M., S. M. Hsiang, and E. Miguel. 2015. "Global Non-Linear Effect of Temperature on Economic Production". *Nature*. 527(7577): 235–39. https://doi.org/10.1038/nature15725.

Byerlee, D., A. D. Janvry, and E. Sadoulet. 2009. "Agriculture for Development: Toward a New Paradigm". *Annual Review of Resource Economics*. 1(1): 15–31. https://doi.org/10.1146/annurev.resource.050708.144239.

Camacho Olmedo, M. T., M. Paegelow, J.-F. Mas, and F. Escobar. 2018. *Geomatic Approaches for Modeling Land Change Scenarios. An Introduction*. Springer.

Carlson, K. M., J. S. Gerber, N. D. Mueller, M. Herrero, G. K. MacDonald, K. A. Brauman, P. Havlik, C. S. O'Connell, J. A. Johnson, S. Saatchi, and P. C. West. 2017. "Greenhouse Gas Emissions Intensity of Global Croplands". *Nature Climate Change*. 7(1): 63–8. https://doi.org/10.1038/nclimate3158.

Carter, C., X. Cui, D. Ghanem, and P. Mérel. 2018. "Identifying the Economic Impacts of Climate Change on Agriculture". *Annual Review of Resource Economics*. 10(1): 361–80. https://doi.org/10.1146/annurev-resource-100517-022938.

Cassman, K. G. and A. Dobermann. 2022. "Nitrogen and the Future of Agriculture: 20 years on: This Article Belongs to Ambio's 50th Anniversary Collection. Theme: Solutions-Oriented Research". *Ambio*. 51(1): 17–24. https://doi.org/10.1007/s13280-021-01526-w.

Chave, J. and C. Thebaud. 1988. "Models of Biodiversity-Mathematical Models". In: *Encyclopedia of Life Support Systems*. Paris, France: Developed under the Auspices of the UNESCO, Eolss Publishers. 400.

Ciais, P., M. Wattenbach, N. Vuichard, P. Smith, S. L. Piao, A. Don, S. Luyssaert, I. A. Janssens, A. Bondeau, R. Dechow, A. Leip, P. Smith, C. Beer, G. R. Van Der Werf, S. Gervois, K. Van Oost, E. Tomelleri, A. Freibauer, E. D. Schulze, and C. S. Team. 2010. "The European Carbon Balance. Part 2: Croplands". *Global Change Biology*. 16(5): 1409–28. https://doi.org/10.1111/j.1365-2486.2009.02055.x.

Colomb, V., S. Ait-Amar, C. Basset-Mens, A. Gac, G. Gaillard, P. Koch, J. Mousset, T. Salou, A. Tailleur, and H. M. Van Der Werf. 2015. "Agribalyse®, the French lci Database for Agricultural Products: High Quality Data for Producers and Environmental Labelling".

Cox, B. 2003. "A Review of Currently Available In-Stream Water-Quality Models and their Applicability for Simulating Dissolved Oxygen in Lowland Rivers". *Science of the Total Environment*. 314: 335–77.

Crabtree, B., W. Earp, and P. Whalley. 1996. "A Demonstration of the Benefits of Integrated Wastewater Planning for Controlling Transient Pollution". *Water Science and Technology*. 33(2): 209–18. https://doi.org/10.2166/wst.1996.0050.

Crippa, M., E. Solazzo, D. Guizzardi, F. Monforti-Ferrario, F. N. Tubiello, and A. Leip. 2021. "Food Systems are Responsible for a Third of Global Anthropogenic Greenhouse Gas Emissions". *Nature Food*. 2(3): 198–209. https://doi.org/10.1038/s43016-021-00225-9.

Crippa, M., E. Solazzo, D. Guizzardi, R. Van Dingenen, and A. Leip. 2022. "Air Pollutant Emissions from Global Food Systems are Responsible for Environmental Impacts, Crop Losses and Mortality". *Nature Food*. 3(11): 942–56. https://doi.org/10.1038/s43016-022-00615-7.

Cui, Y., S. U. Khan, J. Sauer, and M. Zhao. 2022. "Exploring the Spatiotemporal Heterogeneity and Influencing Factors of Agricultural Carbon Footprint and Carbon Footprint Intensity: Embodying Carbon Sink Effect". *Science of The Total Environment*. 846: 157507.

Cunningham, M. A. 2022. "Climate Change, Agriculture, and Biodiversity: How Does Shifting Agriculture Affect Habitat Availability?". *Land*. 11(8): 1257.

Daily, G. C., P. R. Ehrlich, and G. A. Sánchez-Azofeifa. 2001. "Countryside biogeography: Use of human-dominated habitats by the avifauna of southern costa rica". *Ecological Applications*. 11(1): 1–13. https://doi.org/10.1890/1051-0761(2001)011[0001:CBUOHD]2.0.CO;2.

Darwin, R. 1995. *World Agriculture and Climate ChaNGE: ECONOMIC ADAptations*. US Department of Agriculture, Economic Research Service.

Deal, B., V. G. Pallathucheril, Z. Sun, J. Terstriep, and W. Hartel. 2005. "Leam Technical Document: Overview of the LEAM Approach". U.o.I.a. Urbana-Champaign, 76.

Deconinck, K., C. Giner, L. A. Jackson, and L. Toyama. 2021. "Overcoming Evidence Gaps on Food Systems". https://doi.org/10.1787/44ba7574-en.

Delcour, I., P. Spanoghe, and M. Uyttendaele. 2015. "Literature Review: Impact of Climate Change on Pesticide Use". *Food Research International*. 68: 7–15.

Dell, M., B. F. Jones, and B. A. Olken. 2012. "Temperature Shocks and Economic Growth: Evidence from the Last Half Century". *American Economic Journal: Macroeconomics*. 4(3): 66–95. https://doi.org/10.1257/mac.4.3.66.

den Elzen, M., H. Fekete, N. Höhne, A. Admiraal, N. Forsell, A. F. Hof, J. G. Olivier, M. Roelfsema, and H. van Soest. 2016. "Greenhouse Gas Emissions from Current and Enhanced Policies of China Until 2030: Can Emissions Peak before 2030?". *Energy Policy*. 89: 224–36.

Dillon, J. A., K. R. Stackhouse-Lawson, G. J. Thoma, S. A. Gunter, C. A. Rotz, E. Kebreab, D. G. Riley, L. O. Tedeschi, J. Villalba, F. Mitloehner, A. N. Hristov, S. L. Archibeque, J. P. Ritten, and N. D. Mueller. 2021. "Current State of Enteric Methane and the Carbon Footprint of Beef and Dairy Cattle in the United States". *Anim Front*. 11(4): 57–68. https://doi.org/10.1093/af/vfab043.

Doraiswamy, P. C., S. Moulin, P. W. Cook, and A. Stern. 2003. "Crop Yield Assessment from Remote Sensing". *Photogrammetric Engineering & Remote Sensing*. 69(6): 665–74.

Doria, M. D. F., E. Boyd, E. L. Tompkins, and W. N. Adger. 2009. "Using Expert Elicitation to Define Successful Adaptation to Climate Change". *Environmental Science & Policy*. 12(7): 810–9. https://doi.org/10.1016/j.envsci.2009.04.001.

Eady, S. J., T. Grant, H. Cruypenninck, M. Renouf, and G. Mata. 2014. *Ausaglci: A Life Cycle Inventory Database for Australian Agriculture*. RIRDC.

Eckard, R. J., C. Grainger, and C. A. M. de Klein. 2010. "Options for the Abatement of Methane and Nitrous Oxide from Ruminant Production: A Review". *Livestock Science*. 130(1): 47–56. https://doi.org/10.1016/j.livsci.2010.02.010.

Elith, J. and J. R. Leathwick. 2009. "Species Distribution Models: Ecological Explanation and Prediction Across Space and Time". *Annual Review of Ecology, Evolution, and Systematics*. 40(1): 677–97. https://doi.org/10.1146/annurev.ecolsys.110308.120159.

European Commission. 2003. "Common Implementation Strategy for the Water Framework Directive (2000/60/EC)". In: *Policy Summary of Guidance Document No. 23 on Eutrophication Assessment in the Context of European Water Policies*.

Fan, J., C. Liu, J. Xie, L. Han, C. Zhang, D. Guo, J. Niu, H. Jin, and B. G. McConkey. 2022. "Life Cycle Assessment on Agricultural Production: A Mini Review on Methodology, Application, and Challenges". *International Journal of Environirонmental Research & Public Health*. 19(16). https://doi.org/10.3390/ijerph19169817.

Fankhauser, S. 2017. "Adaptation to Climate Change". *Annual Review of Resource Economics*. 9(1): 209–30. https://doi.org/10.1146/annurev-resource-100516-033554.

Forsell, N., O. Turkovska, M. Gusti, M. Obersteiner, M. D. Elzen, and P. Havlik. 2016. "Assessing the INDCS' Land Use, Land Use Change, and Forest Emission Projections". *Carbon Balance and Management*. 11(1): 1–17.

Fowler, D., M. Coyle, U. Skiba, M. A. Sutton, J. N. Cape, S. Reis, L. J. Sheppard, A. Jenkins, B. Grizzetti, J. N. Galloway, P. Vitousek, A. Leach, A. F. Bouwman, K. Butterbach-Bahl, F. Dentener, D. Stevenson, M. Amann, and M. Voss. 2013. "The Global Nitrogen Cycle in the Twenty-First Century". *Philosophical Transactions of the Royal Society B: Biological Sciences*. 368(1621): 20130164. https://doi.org/10.1098/rstb.2013.0164.

Gallardo, K. and Q. Wang. 2013. "Willingness to Pay for Pesticides' Environmental Features and Social Desirability Bias: The Case of Apple and Pear Growers". *Journal of Agricultural and Resource Economics*. 38: 124–39. https://doi.org/10.22004/ag.econ.148250.

Geng, A., H. Yang, J. Chen, and Y. Hong. 2017. "Review of Carbon Storage Function of Harvested Wood Products and the Potential of Wood Substitution in Greenhouse Gas Mitigation". *Forest Policy and Economics*. 85: 192–200.

Getson, J. M., S. P. Church, B. G. Radulski, A. E. Sjöstrand, J. Lu, and L. S. Prokopy. 2022. "Understanding Scientists' Communication Challenges at the Intersection of Climate and Agriculture". *PLOS ONE*. 17(8): e0269927. https://doi.org/10.1371/journal.pone.0269927.

Ghosh, P. and S. P. Kumpatla. 2022. "GIS Applications in Agriculture". In: *Geographic Information Systems and Applications in Coastal Studies*. IntechOpen.

Giri, C. P. 2012. *Remote Sensing of Land Use and Land Cover: Principles and Applications*. CRC press.

Glidden, C. K., N. Nova, M. P. Kain, K. M. Lagerstrom, E. B. Skinner, L. Mandle, S. H. Sokolow, R. K. Plowright, R. Dirzo, and G. A. De Leo. 2021. "Human-Mediated Impacts on Biodiversity and the Consequences for Zoonotic Disease Spillover". *Current Biology*. 31(19): R1342–R1361.

Gonzalez-Quiñones, V., D. Murphy, R. Bowles, and P. Mele. 2015. "A National Soil Quality Monitoring Framework". *GRDC Soil Biology Initiative II. Final Report. UWA000138*: 258.

González-Ramírez, J., C. L. Kling, and A. Valcu. 2012. "An Overview of Carbon Offsets from Agriculture". *Annual Review of Resource Economics*. 4(1): 145–60. https://doi.org/10.1146/annurev-resource-083110-120016.

Grassi, G., J. House, F. Dentener, S. Federici, M. den Elzen, and J. Penman. 2017. "The Key Role of Forests in Meeting Climate Targets Requires Science for Credible Mitigation". *Nature Climate Change*. 7(3): 220–26. https://doi.org/10.1038/nclimate3227.

Haas, G., F. Wetterich, and U. Geier. 2000. "Life Cycle Assessment Framework in Agriculture on the Farm Level". *The International Journal of Life Cycle Assessment*. 5: 345–48.

Hanley, N. and C. Perrings. 2019. "The Economic Value of Biodiversity". *Annual Review of Resource Economics*. 11(1): 355–75. https://doi.org/10.1146/annurev-resource-100518-093946.

Harmel, R. D., P. K. Smith, K. W. Migliaccio, I. Chaubey, K. R. Douglas-Mankin, B. Benham, S. Shukla, R. Muñoz-Carpena, and B. J. Robson. 2014. "Evaluating, Interpreting, and Communicating Performance of Hydrologic/Water Quality Models Considering Intended Use: A Review and Recommendations". *Environmental Modelling & Software*. 57: 40–51. https://doi.org/10.1016/j.envsoft.2014.02.013.

Houghton, R. A. and A. A. Nassikas. 2018. "Negative Emissions from Stopping Deforestation and Forest Degradation, Globally". *Global Change Biology*. 24(1): 350–59. https://doi.org/10.1111/gcb.13876.

Huber, S., B. Syed, A. Freudenschuss, V. Ernstsen, and P. Loveland. 2001. *Proposal for A European Soil Monitoring and Assessment Framework*. EEA.

Ibidhi, R., T.-H. Kim, R. Bharanidharan, H.-J. Lee, Y.-K. Lee, N.-Y. Kim, and K.-H. Kim. 2021. "Developing Country-specific Methane Emission Factors and Carbon Fluxes from Enteric Fermentation in South Korean Dairy Cattle Production". *Sustainability*. 13(16): 9133.

Idowu, O. J., H. M. van Es, G. S. Abawi, D. W. Wolfe, J. I. Ball, B. K. Gugino, B. N. Moebius, R. R. Schindelbeck, and A. V. Bilgili. 2008. "Farmer-oriented Assessment of Soil Quality using Field, Laboratory, and VNIR Spectroscopy Methods". *Plant and Soil*. 307(1): 243–53. https://doi.org/10.1007/s11104-007-9521-0.

Ierodiaconou, D., L. Laurenson, M. Leblanc, F. Stagnitti, G. Duff, S. Salzman, and V. Versace. 2005. "The Consequences of Land Use Change on Nutrient Exports: A Regional Scale Assessment in South-West Victoria, Australia". *Journal of Environmental Management*. 74(4): 305–16. https://doi.org/10.1016/j.jenvman.2004.09.010.

Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. 2019. *Modelling Impacts of Drivers on Biodiversity and Ecosystems*. https://www.ipbes.net/modelling-impacts-drivers-biodiversity-ecosystems#:~:text=Models%20of%20biodiversity%20and%20ecosystem,level%20models%20and%20evolutionary%20adaptation.

IPCC. 2014. "Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part A: Global and Sectoral Aspects". *Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*.

IPCC. 2019. "Climate Change and Land: An IPCC Special Report on Climate Change Desertification, Land Degradation, Sustainable Land Management, Food Security, and Greenhouse Gas Fluxes in Terrestrial Ecosystems". In: ed. by P. R. Shukla, J. Skea, E. C. Buendia, V. Masson-Delmotte, H. O. Pörtner, D. C. Roberts, P. Zhai, R. Slade, S. Connors, R. V. Diemen, M. Ferrat, E. Haughey, S. Luz, S. Neogi, M. Pathak, J. Petzold, J. P. Pereira, P. Vyas, E. Huntley, K. Kissick, M. Belkacemi, and J. Malley. In press.

ISO. 2006a. ISO (14040). *Environmental Management — Life Cycle Assessment — Principles and Framework* (ISO 14040: 2006). Geneva, Switzerland: International Standards Organization.

ISO. 2006b. ISO (14044). *Environmental Management — Life Cycle Assessment — Requirements and Management*. Geneva, Switzerland: International Standards Organization.

Johnes, P. J. 1996. "Evaluation and Management of the Impact of Land Use Change on the Nitrogen and Phosphorus Load Delivered to Surface Waters: The Export Coefficient Modelling Approach". *Journal of Hydrology*. 183(3): 323–49. https://doi.org/10.1016/0022-1694(95)02951-6.

Jones, J. W., J. M. Antle, B. Basso, K. J. Boote, R. T. Conant, I. Foster, H. C. J. Godfray, M. Herrero, R. E. Howitt, and S. Janssen. 2017. "Brief History of Agricultural Systems Modeling". *Agricultural Systems*. 155: 240–54.

Jørgensen, S. E., H. Löffler, W. Rast, and M. Straškraba. 2005. "The Use of Mathematical Modelling in Lake and Reservoir Management". In: *Developments in Water Science*. Oxford: Elsevier. 243–314.

Karlen, D. L., S. S. Andrews, and J. W. Doran. 2001. "Soil Quality: Current Concepts and Applications". *Advances in Agronomy*, Academic Press, pp. 1–40.

Kaur, R., G. K. Mavi, S. Raghav, and I. Khan. 2019. "Pesticides Classification and Its Impact on Environment". *International Journal of Current Microbiology and Applied Science*. 8(3): 1889–97.

Kazmi, A. A. and I. S. Hansen. 1997. "Numerical Models in Water Quality Management: A Case Study for the Yamuna River (India)". *Water Science and Technology*. 36(5): 193–200.

Kearney, M. and W. Porter. 2009. "Mechanistic Niche Modelling: Combining Physiological and Spatial Data to Predict Species' Ranges". *Ecology Letters*. 12(4): 334–50. https://doi.org/10.1111/j.1461-0248.2008.01277.x.

Kehoe, L., A. Romero-Muñoz, E. Polaina, L. Estes, H. Kreft, and T. Kuemmerle. 2017. "Biodiversity at Risk under Future Cropland Expansion and Intensification". *Nature Ecology & Evolution*. 1(8): 1129–35. https://doi.org/10.1038/s41559-017-0234-3.

Khajuria, A. and N. Ravindranath. 2012. "Climate Change Vulnerability Assessment: Approaches DPSIR Framework and Vulnerability Index". *Journal of Earth Science & Climatic Change*. 3: 109.

Khanal, S., J. Fulton, E. Hawkins, K. Port, and A. Klopfenstein. 2017. *Remote Sensing in Precision Agriculture*. https://ohioline.osu.edu/factsheet/fabe-5541.

Kim, J. B., E. Monier, B. Sohngen, G. S. Pitts, R. Drapek, J. McFarland, S. Ohrel, and J. Cole. 2017. "Assessing Climate Change Impacts, Benefits of Mitigation, and Uncertainties on Major Global Forest Regions under Multiple Socioeconomic and Emissions Scenarios". *Environmental Research Letters*. 12(4): 045001.

Klöpffer, W. 1997. "Life Cycle Assessment". *Environmental Science and Pollution Research*. 4(4): 223–28. https://doi.org/10.1007/BF02986351.

Kroes, J., P. Groenendijk, D. de Abelleyra, S. R. Verón, D. Plotnikov, S. Bartalev, N. Yan, B. Wu, N. Kussul, and S. Fritz. 2017. *Environmental Impact Assessment of Agricultural Land Use Changes*. S.I.f.G.M.o.A. (SIGMA).

Kross, A., G. Kaur, and J. A. G. Jaeger. 2022. "A Geospatial Framework for the Assessment and Monitoring of Environmental Impacts of Agriculture". *Environmental Impact Assessment Review*. 97: 106851. https://doi.org/10.1016/j.eiar.2022.106851.

Kumaş, K. and A. Ö. Akyüz. 2023. "Estimation of Greenhouse Gas Emission and Global Warming Potential of Livestock Sector; Lake District, Türkiye". *International Journal of Environment and Geoinformatics*. 10(1): 132–38.

Kurukulasuriya, P. and R. Mendelsohn. 2007. "Crop Selection: Adapting to Climage Change in Africa". *Crop Selection: Adapting To Climage Change In Africa*. https://elibrary.worldbank.org/doi/abs/10.1596/1813-9450-4307 (last accessed).

Lamichhane, J. R. 2017. "Pesticide Use and Risk Reduction in European Farming Systems with IPM: An Introduction to the Special Issue". *Crop Protection*. 97: 1–6.

Lawell, C.-Y. C. L. 2016. "The Management of Groundwater: Irrigation Efficiency, Policy, Institutions, and Externalities". *Annual Review of Resource Economics*. 8(1): 247–59. https://doi.org/10.1146/annurev-resource-100815-095425.

Lechenet, M., F. Dessaint, G. Py, D. Makowski, and N. Munier-Jolain. 2017. "Reducing Pesticide Use While Preserving Crop Productivity and Profitability on Arable Farms". *Nature Plants*. 3(3): 1–6.

Lemly, A. D., R. T. Kingsford, and J. R. Thompson. 2000. "Irrigated Agriculture and Wildlife Conservation: Conflict on a Global Scale". *Environmental Management*. 25(5): 485–512.

Lilburne, L., G. Sparling, and L. Schipper. 2004. "Soil Quality Monitoring in New Zealand: Development of an Interpretative Framework". *Agriculture, Ecosystems & Environment*. 104(3): 535–44. https://doi.org/10.1016/j.agee.2004.01.020.

Loveland, P. and T. Thompson. 2002. *Identification and Development of a Set of National Indicators for Soil Quality*. Environment Agency.

Lyon, J. G. 2002. *GIS for Water Resource and Watershed Management*. CRC Press.

MacDonald, K., F. Wang, W. Fraser, and G. Lelyk. 1998. "Broad-scale Assessment of Agricultural Soil Quality in Canada Using Exiting Land Resource Databases and GIS. Research Branch Tech". *Bull.*

Mace, G. M., K. Norris, and A. H. Fitter. 2012. "Biodiversity and Ecosystem Services: A Multilayered Relationship". *Trends in Ecology & Evolution*. 27(1): 19–26. https://doi.org/10.1016/j.tree.2011.08.006.

Mahmood, I., S. R. Imadi, K. Shazadi, A. Gul, and K. R. Hakeem. 2016. "Effects of Pesticides on Environment". *Plant, Soil and Microbes: Volume 1: Implications in Crop Science*: 253–69.

Malhi, G. S., M. Kaur, and P. Kaushik. 2021. "Impact of Climate Change on Agriculture and its Mitigation Strategies: A Review". *Sustainability*. 13(3): 1318.

Malomo, G. A., A. S. Madugu, and S. A. Bolu. 2018. "Sustainable Animal Manure Management Strategies and Practices". In: *Agricultural Waste and Residues*. Ed. by A. Anna. Rijeka: IntechOpen, Ch. 8.

Martin, A. E., S. J. Collins, S. Crowe, J. Girard, I. Naujokaitis-Lewis, A. C. Smith, K. Lindsay, S. Mitchell, and L. Fahrig. 2020. "Effects of Farmland Heterogeneity on Biodiversity are Similar To — or Even Larger than — The Effects of Farming Practices". *Agriculture, Ecosystems & Environment*. 288: 106698.

Martin, S., D. Baize, M. Bonneau, R. Chaussod, J. Gaultier, P. Lavelle, J. Legros, A. Leprêtre, and T. Sterckeman. 1998. "The French National Soil Quality Observatory". In: *Proceedings of the 16th World Congress on Soil Science, Symposium*. 20–6.

Mateo, R. G., K. Mokany, and A. Guisan. 2017. "Biodiversity Models: What If Unsaturation is the Rule?". *Trends in Ecology & Evolution*. 32(8): 556–66.

McCarl, B. A. 2007. "Adaptation Options for Agriculture, Forestry and Fisheries. A Report to the UNFCC Secretariat Financial and Technical Support Division". In: *United Nations Framework Convention on Climate Change*.

Mekonnen, M. M. and A. Y. Hoekstra. 2015. "Global Gray Water Footprint and Water Pollution Levels Related to Anthropogenic Nitrogen Loads to Fresh Water". *Environmental Science & Technology*. 49(21): 12860–868. https://doi.org/10.1021/acs.est.5b03191.

Mendelsohn, R. and A. Dinar. 2009. "Land Use and Climate Change Interactions". *Annual Review of Resource Economics*. 1(1): 309–32. https://doi.org/10.1146/annurev.resource.050708.144246.

Mendelsohn, R., I. C. Prentice, O. Schmitz, B. Stocker, R. Buchkowski, and B. Dawson. 2016. "The Ecosystem Impacts of Severe Warming". *American Economic Review*. 106(5): 612–14. https://doi.org/10.1257/aer.p20161104.

Merrington, G., S. Fishwick, D. Barraclough, J. Morris, N. Preedy, T. Boucard, M. Reeve, P. Smith, and C. Fang. 2006. "The Development and Use of Soil Quality Indicators for Assessing the Role of Soil in Environmental Interactions". *Report to UK SIC, SC030265, Environment Agency*.

Metaxoglou, K. and S. Aaron. 2022. "Nutrient Pollution and U.S. Agriculture: Causal Effects, Integrated Assessment, and Implications of Climate Change". *National Bureau of Economic Research*, Working Paper (30124) https://doi.org/10.3386/w30124.

Min, B. R., S. Lee, H. Jung, D. N. Miller, and R. Chen. 2022. "Enteric Methane Emissions and Animal Performance in Dairy and Beef Cattle Production: Strategies, Opportunities, and Impact of Reducing Emissions". *Animals (Basel)*. 12(8). https://doi.org/10.3390/ani12080948.

Moebius-Clune, B. N. 2016. *Comprehensive Assessment of Soil Health: The Cornell Framework Manual*. Cornell University.

Momtaz, S. 2002. "Environmental Impact Assessment in Bangladesh: A Critical Review". *Environmental Impact Assessment Review*. 22(2): 163–79. https://doi.org/10.1016/S0195-9255(01)00106-8.

Montes, F., R. Meinen, C. Dell, A. Rotz, A. N. Hristov, J. Oh, G. Waghorn, P. J. Gerber, B. Henderson, H. P. Makkar, and J. Dijkstra. 2013. "Special Topics–Mitigation of Methane and Nitrous Oxide Emissions from Animal Operations: II. A Review of Manure Management Mitigation Options". *Journal of Animal Science*. 91(11): 5070–94. https://doi.org/10.2527/jas.2013-6584.

Moriasi, D. N., M. W. Gitau, N. Pai, and P. Daggupati. 2015. "Hydrologic and Water Quality Models: Performance Measures and Evaluation Criteria". *Transactions of the ASABE*. 58(6): 1763–785.

Münier, B., K. Birr-Pedersen, and J. Schou. 2004. "Combined Ecological and Economic Modelling in Agricultural Land Use Scenarios". *Ecological Modelling*. 174(1–2): 5–18.

Nabuurs, G.-J., P. Delacote, D. Ellison, M. Hanewinkel, L. Hetemäki, and M. Lindner. 2017. "By 2050 the Mitigation Effects of EU Forests Could Nearly Double through Climate Smart Forestry". *Forests*. 8(12): 484.

Navarro, J., B. A. Bryan, O. Marinoni, S. Eady, and A. Halog. 2016. "Mapping Agriculture's Impact by Combining Farm Management Handbooks, Life-Cycle Assessment and Search Engine Science". *Environmental Modelling & Software*. 80: 54–65. https://doi.org/10.1016/j.envsoft.2016.02.020.

Nelson, G. C., H. Valin, R. D. Sands, P. Havlík, H. Ahammad, D. Deryng, J. Elliott, S. Fujimori, T. Hasegawa, and E. Heyhoe. 2014. "Climate Change Effects on Agriculture: Economic Responses to Biophysical Shocks". *Proceedings of the National Academy of Sciences*. 111(9): 3274–79.

Neue, H.-U. 1993a. "Methane Emission from Rice Fields". *BioScience*. 43(7): 466–74. https://doi.org/10.2307/1311906.

Neue, H.-U. 1993b. “Methane Emission from Rice Fields: Wetland Rice Fields May Make A Major Contribution to Global Warming”. *BioScience*. 43(7): 466–74. https://doi.org/10.2307/1311906.

Outhwaite, C. L., P. McCann, and T. Newbold. 2022. “Agriculture and Climate Change Are Reshaping Insect Biodiversity Worldwide”. *Nature*. 605(7908): 97–102. https://doi.org/10.1038/s41586-022-04644-x.

Pandey, D. and M. Agrawal. 2014. “Carbon Footprint Estimation in the Agriculture Sector”. In: *Assessment of Carbon Footprint in Different Industrial Sectors, Volume 1*. Ed. by S. S. Muthu. Singapore: Springer Singapore. 25–47.

Parthasarathi, T., K. Vanitha, S. Mohandass, and E. Vered. 2019. “Mitigation of Methane Gas Emission in Rice by Drip Irrigation [Version 1; Peer Review: 2 Approved]”. *F1000Research*. 8(2023). https://doi.org/10.12688/f1000research.20945.1.

Paul, C., N. Hanley, S. T. Meyer, C. Fürst, W. W. Weisser, and T. Knoke. 2020. “On the Functional Relationship between Biodiversity and Economic Value”. *Science Advances*. 6(5): eaax7712. https://doi.org/10.1126/sciadv.aax7712.

Payraudeau, S. and H. M. G. van der Werf. 2005. “Environmental Impact Assessment for a Farming Region: A Review of Methods”. *Agriculture, Ecosystems & Environment*. 107(1): 1–19. https://doi.org/10.1016/j.agee.2004.12.012.

Pereira, H. M. and G. C. Daily. 2006. “Modeling Biodiversity Dynamics in Countryside Landscapes”. *Ecology*. 87(8): 1877–85.

Pereira, H. M., P. W. Leadley, V. Proença, R. Alkemade, J. P. Scharlemann, J. F. Fernandez-Manjarrés, M. B. Araújo, P. Balvanera, R. Biggs, and W. W. Cheung. 2010. “Scenarios for Global Biodiversity in the 21st Century”. *Science*. 330(6010): 1496–501.

Pérez-Lucas, G., N. Vela, A. El Aatik, and S. Navarro. 2019. “Environmental Risk of Groundwater Pollution by Pesticide Leaching through the Soil Profile”. *Pesticides-Use and Misuse and Their Impact in the Environment*: 1–28.

Perminova, T., N. Sirina, B. Laratte, N. Baranovskaya, and L. Rikhvanov. 2016. “Methods for Land Use Impact Assessment: A Review”. *Environmental Impact Assessment Review*. 60: 64–74. https://doi.org/10.1016/j.eiar.2016.02.002.

Peter, C., K. Helming, and C. Nendel. 2017. "Do Greenhouse Gas Emission Calculations from Energy Crop Cultivation Reflect Actual Agricultural Management Practices? — A Review of Carbon Footprint Calculators". *Renewable and Sustainable Energy Reviews*. 67: 461–76.

Pierce, F. J. and D. Clay. 2007. *GIS Applications in Agriculture*. CRC Press.

Potter, P., N. Ramankutty, E. M. Bennett, and S. D. Donner. 2010. "Characterizing the Spatial Patterns of Global Fertilizer Application and Manure Production". *Earth Interactions*. https://doi.org/10.1175/2010ei288.1.

Pushpalatha, R., C. Perrin, N. L. Moine, and V. Andréassian. 2012. "A Review of Efficiency Criteria Suitable for Evaluating Low-flow Simulations". *Journal of Hydrology*. 420–421: 171–82. https://doi.org/10.1016/j.jhydrol.2011.11.055.

Rasche, L., J. C. Habel, N. Stork, E. Schmid, and U. A. Schneider. 2022. "Food Versus Wildlife: Will Biodiversity Hotspots Benefit from Healthier Diets?". *Global Ecology and Biogeography*. 31(6): 1090–103.

Rauh, E. 2021. "Exploring Intensive Agriculture and Organic Fertilizer Management in the U.S.: Implications for Nutrient Pollution Prevention". In: *School of Sustainable Enginnering*. Tempe, AZ: Arizona State University.

Rebolledo-Leiva, R., L. Angulo-Meza, A. Iriarte, and M. C. González-Araya. 2017. "Joint Carbon Footprint Assessment and Data Envelopment Analysis for the Reduction of Greenhouse Gas Emissions in Agriculture Production". *Science of the Total Environment*. 593: 36–46.

Reilly, J., N. Hohmann, and S. Kane. 1994. "Climate Change and Agricultural Trade: Who Benefits, Who Loses?". *Global Environmental Change*. 4(1): 24–36.

Rey Benayas, J. M. and J. M. Bullock. 2015. "Vegetation Restoration and Other Actions to Enhance Wildlife in European Agricultural Landscapes". In: *Rewilding European Landscapes*. Springer International Publishing Cham. 127–42.

Rinot, O., G. J. Levy, Y. Steinberger, T. Svoray, and G. Eshel. 2019. "Soil Health Assessment: A Critical Review of Current Methodologies and a Proposed New Approach". *Science of the Total Environment*. 648: 1484–91.

Ritter, A. and R. Muñoz-Carpena. 2013. "Performance Evaluation of Hydrological Models: Statistical Significance for Reducing Subjectivity in Goodness-of-Fit Assessments". *Journal of Hydrology*. 480: 33–45. https://doi.org/10.1016/j.jhydrol.2012.12.004.

Rivera-Huerta, A., M. de la Salud Rubio Lozano, J. C. Ku-Vera, and L. P. Güereca. 2022. "Emission Factors from Enteric Fermentation of Different Categories of Cattle in the Mexican Tropics: A Comparison between 2006 and 2019 IPCC". *Climatic Change*. 172(3–4): 23.

Rodrigues, G. S., C. Campanhola, and P. C. Kitamura. 2003. "An Environmental Impact Assessment System for Agricultural R&D". *Environmental Impact Assessment Review*. 23(2): 219–44.

Rondinini, C. and P. Visconti. 2015. "Scenarios of Large Mammal Loss in Europe for the 21st Century". *Conservation Biology*. 29(4): 1028–36. https://doi.org/10.1111/cobi.12532.

Rosegrant, M. W., C. Ringler, and T. Zhu. 2009. "Water for Agriculture: Maintaining Food Security under Growing Scarcity". *Annual Review of Environment and Resources*. 34(1): 205–22. https://doi.org/10.1146/annurev.environ.030308.090351.

Rosenzweig, C. and M. L. Parry. 1994. "Potential Impact of Climate Change on World Food Supply". *Nature*. 367(6459): 133–38.

Sala, O. E., J. J. A. F. Stuart Chapin, III, E. Berlow, J. Bloomfield, R. Dirzo, E. Huber-Sanwald, L. F. Huenneke, R. B. Jackson, A. Kinzig, R. Leemans, D. M. Lodge, H. A. Mooney, M. N. Oesterheld, N. L. Poff, M. T. Sykes, B. H. Walker, M. Walker, and D. H. Wall. 2000. "Global Biodiversity Scenarios for the Year 2100". *Science*. 287(5459): 1770–74. DOI: https://doi.org/10.1126/science.287.5459.1770.

Salem, B. 2003. "Application of GIS to Biodiversity Monitoring". *Journal of Arid Environments*. 54(1): 91–114.

Schipper, L. A. and G. P. Sparling. 2000. "Performance of Soil Condition Indicators Across Taxonomic Groups and Land Uses". *Soil Science Society of America Journal*. 64(1): 300–11. https://doi.org/10.2136/sssaj2000.641300x.

Segerson, K. 2013. "Voluntary Approaches to Environmental Protection and Resource Management". *Annual Review of Resource Economics*. 5(1): 161–80. https://doi.org/10.1146/annurev-resource-091912-151945.

Segerson, K. 2022. "Group Incentives for Environmental Protection and Natural Resource Management". *Annual Review of Resource Economics*. 14(1): 597–619. https://doi.org/10.1146/annurev-resource-111920-020235.

Seo, N. S. and R. Mendelsohn. 2008. "A Ricardian Analysis of the Impact of Climate Change on South American Farms". *Chilean Journal of Agricultural Research*. 68: 69–79.

Sequeira, A. M., P. J. Bouchet, K. L. Yates, K. Mengersen, and M. J. Caley. 2018. "Transferring Biodiversity Models for Conservation: Opportunities and Challenges". *Methods in Ecology and Evolution*. 9(5): 1250–64.

Shanmugapriya, P., S. Rathika, T. Ramesh, and P. Janaki. 2019. "Applications of Remote Sensing in Agriculture — A Review". *International Journal of Current Microbiology and Applied Sciences*. 8(01): 2270–83.

Sharma, A., V. Kumar, B. Shahzad, M. Tanveer, G. P. S. Sidhu, N. Handa, S. K. Kohli, P. Yadav, A. S. Bali, R. D. Parihar, O. I. Dar, K. Singh, S. Jasrotia, P. Bakshi, M. Ramakrishnan, S. Kumar, R. Bhardwaj, and A. K. Thukral. 2019. "Worldwide Pesticide Usage and its Impacts on Ecosystem". *SN Applied Sciences*. 1(11): 1446. https://doi.org/10.1007/s42452-019-1485-1.

Sobota, D. J., J. E. Compton, M. L. McCrackin, and S. Singh. 2015. "Cost of Reactive Nitrogen Release from Human Activities to the Environment in the United States". *Environmental Research Letters*. 10(2): 025006. https://doi.org/10.1088/1748-9326/10/2/025006.

Sohngen, B. 2020. "Climate Change and Forests". *Annual Review of Resource Economics*. 12(1): 23–43. https://doi.org/10.1146/annurev-resource-110419-010208.

Sommer, S. G., J. E. Olesen, S. O. Petersen, M. R. Weisbjerg, L. Valli, L. Rodhe, and F. Beline. 2009. "Region-specific Assessment of Greenhouse Gas Mitigation with Different Manure Management Strategies in Four Agroecological Zones". *Global Change Biology*. 15(12): 2825–37. https://doi.org/10.1111/j.1365-2486.2009.01888.x.

Sparling, G. P., L. A. Schipper, W. Bettjeman, and R. Hill. 2004. "Soil Quality Monitoring in New Zealand: Practical Lessons from a 6-year Trial". *Agriculture, Ecosystems & Environment*. 104(3): 523–34. https://doi.org/10.1016/j.agee.2004.01.021.

Sparling, G. and L. Schipper. 2004. "Soil Quality Monitoring in New Zealand: Trends and Issues Arising from a Broad-Scale Survey". *Agriculture, Ecosystems & Environment*. 104(3): 545–52. https://doi.org/10.1016/j.agee.2003.11.014.

Steinfeld, C. M., A. Sharma, R. Mehrotra, and R. T. Kingsford. 2020. "The Human Dimension of Water Availability: Influence of Management Rules on Water Supply for Irrigated Agriculture and the Environment". *Journal of Hydrology*. 588: 125009.

Stephenson, K. and L. Shabman. 2017. "Can Water Quality Trading Fix the Agricultural Nonpoint Source Problem?". *Annual Review of Resource Economics*. 9(1): 95–116. https://doi.org/10.1146/annurev-resource-100516-053639.

Stern, N. 2013. "The Structure of Economic Modeling of the Potential Impacts of Climate Change: Grafting Gross Underestimation of Risk onto Already Narrow Science Models". *Journal of Economic Literature*. 51(3): 838–59.

Stevens, A. W. 2018. "The Economics of Soil Health". *Food Policy*. 80: 1–9.

Tallis, H., T. Ricketts, E. Nelson, D. Ennaanay, S. Wolny, N. Olwero, K. Vigerstol, D. Pennington, G. Mendoza, and J. Aukema. 2010. *Invest 1.004 Beta User's Guide*. The Natural Capital Project: Stanford, CA, USA.

Tan, G. and R. Shibasaki. 2003. "Global Estimation of Crop Productivity and the Impacts of Global Warming by GIS and Epic Integration". *Ecological Modelling*. 168(3): 357–70.

Tang, S., L. Ma, X. Wei, D. Tian, B. Wang, Z. Li, Y. Zhang, and X. Shao. 2019. "Methane Emissions in Grazing Systems in Grassland Regions of China: A Synthesis". *Science of the Total Environment*. 654: 662–70. https://doi.org/10.1016/j.scitotenv.2018.11.102.

Thakuri, S., P. Baskota, S. B. Khatri, A. Dhakal, P. Chaudhary, K. Rijal, and R. M. Byanju. 2020. "Methane Emission Factors and Carbon Fluxes from Enteric Fermentation in Cattle of Nepal Himalaya". *Science of The Total Environment*. 746: 141184.

Thauer, R. K., A.-K. Kaster, H. Seedorf, W. Buckel, and R. Hedderich. 2008. "Methanogenic Archaea: Ecologically Relevant Differences in Energy Conservation". *Nature Reviews Microbiology*. 6(8): 579–91. https://doi.org/10.1038/nrmicro1931.

Tongwane, M. I. and M. E. Moeletsi. 2020. "Emission Factors and Carbon Emissions of Methane from Enteric Fermentation of Cattle Produced under Different Management Systems in South Africa". *Journal of Cleaner Production*. 265: 121931.

Tsakiris, G. and D. Alexakis. 2012. "Water Quality Models: An Overview". *European Water*. 37: 33–46.

Tscharntke, T., Y. Clough, T. C. Wanger, L. Jackson, I. Motzke, I. Perfecto, J. Vandermeer, and A. Whitbread. 2012. "Global Food Security, Biodiversity Conservation and the Future of Agricultural Intensification". *Biological Conservation*. 151(1): 53–9.

Tubiello, F. N., M. Salvatore, A. F. Ferrara, J. House, S. Federici, S. Rossi, R. Biancalani, R. D. Condor Golec, H. Jacobs, A. Flammini, P. Prosperi, P. Cardenas-Galindo, J. Schmidhuber, M. J. Sanz Sanchez, N. Srivastava, and P. Smith. 2015. "The Contribution of Agriculture, Forestry and Other Land Use Activities to Global Warming, 1990–2012". *Global Change Biology*. 21(7): 2655–60. https://doi.org/10.1111/gcb.12865.

Tudi, M., H. Daniel Ruan, L. Wang, J. Lyu, R. Sadler, D. Connell, C. Chu, and D. T. Phung. 2021. "Agriculture Development, Pesticide Application and its Impact on the Environment". *International Journal of Environironmental Research & Public Health*. 18(3). https://doi.org/10.3390/ijerph18031112.

Turcotte, M. M., H. Araki, D. S. Karp, K. Poveda, and S. R. Whitehead. 2017. "The Eco-evolutionary Impacts of Domestication and Agricultural Practices on Wild Species". *Philosophical Transactions of the Royal Society B: Biological Sciences*. 372(1712): 20160033. https://doi.org/10.1098/rstb.2016.0033.

U.N. Food and Agriculture Organization. 2014. *Agriculture, Forestry and Other Land Use Emissions bby Sources and Renewals by Sinks: 1990–2011 Analysis*. F.S.D.W.P.S. 14/01. Rome, Italy.

U.N. Food and Agriculture Organization. 2016. *The State of Food and Agriculture 2016." Climate Change, Agriculture, and Food Security*. Italy, Rome.

U.N. Food and Agriculture Organization. 2018. *The Future of Food and Agriculture. Alternative Pathways to 2050*. Rome, Italy.

U.N. Food and Agriculture Organization. 2020. *Faostat Agri-environmental Indicators, Emission Shares*. Italy, Rome.

U.N. Food and Agriculture Organization. 2021. *Nutrition. Food Loss and Waste*. Rome, Italy.

UNESCO World Water Assessment Programme. 2019. *The United Nations World Water Development Report 2019: Leaving No One Behnid*. Paris, France.

U.S. Environmental Protection Agency. 2022. *Inventory of U.S. Greenhouse Gas Emissions and Sinks*.

van Ittersum, M. K., F. Ewert, T. Heckelei, J. Wery, J. Alkan Olsson, E. Andersen, I. Bezlepkina, F. Brouwer, M. Donatelli, G. Flichman, L. Olsson, A. E. Rizzoli, T. van der Wal, J. E. Wien, and J. Wolf. 2008. "Integrated Assessment of Agricultural Systems — A Component-based Framework for the European Union (seamless)". *Agricultural Systems*. 96(1): 150–65. https://doi.org/10.1016/j.agsy.2007.07.009.

Van Vuuren, D. P., J. Edmonds, M. Kainuma, K. Riahi, A. Thomson, K. Hibbard, G. C. Hurtt, T. Kram, V. Krey, J.-F. Lamarque, T. Masui, M. Meinshausen, N. Nakicenovic, S. J. Smith, and S. K. Rose. 2011. "The Representative Concentration Pathways: An Overview". *Climatic Change*. 109(1): 5. https://doi.org/10.1007/s10584-011-0148-z.

Veerkamp, C. J., M. Loreti, R. Benavidez, B. Jackson, and A. M. Schipper. 2023. "Comparing Three Spatial Modeling Tools for Assessing Urban Ecosystem Services". *Ecosystem Services*. 59(101500). https://doi.org/10.1016/j.ecoser.2022.101500.

Veldkamp, A. and L. Fresco. 1996. "Clue: A Conceptual Model to Study the Conversion of Land Use and Its Effects". *Ecological Modelling*. 85(2–3): 253–70.

Venohr, M., U. Hirt, J. Hofmann, D. Opitz, A. Gericke, A. Wetzig, S. Natho, F. Neumann, J. Hürdler, and M. Matranga. 2011. "Modelling of Nutrient Emissions in River Systems–Moneris–Methods and Background". *International Review of Hydrobiology*. 96(5): 435–83.

Verburg, P. H. and K. P. Overmars. 2009. "Combining Top-down and Bottom-up Dynamics in Land Use Modeling: Exploring the Future of Abandoned Farmlands in Europe with the Dyna-clue Model". *Landscape Ecology*. 24: 1167–81.

Vilas, M. P., P. J. Thorburn, S. Fielke, T. Webster, M. Mooij, J. S. Biggs, Y.-F. Zhang, A. Adham, A. Davis, B. Dungan, R. Butler, and P. Fitch. 2020. "1622wq: A Web-based Application to Increase Farmer Awareness of the Impact of Agriculture on Water Quality". *Environmental Modelling & Software*. 132: 104816. https://doi.org/10.1016/j.envsoft.2020.104816.

Wang, C., B. D. Walker, and H. W. Rees. 1997. "Chapter 15 Establishing a Benchmark System for Monitoring Soil Quality in Canada". In: *Developments in Soil Science*. Ed. by E. G. Gregorich and M. R. Carter. 323–37.

Wang, J., R. Mendelsohn, A. Dinar, J. Huang, S. Rozelle, and L. Zhang. 2009. "The Impact of Climate Change on China's Agriculture". *Agricultural Economics*. 40(3): 323–37.

Wang, Q., S. Li, P. Jia, C. Qi, and F. Ding. 2013. "A Review of Surface Water Quality Models". *The Scientific World Journal*. 2013: 1–7. https://doi.org/10.1155/2013/231768.

Ward, F. A. 2022. "Enhancing Climate Resilience of Irrigated Agriculture: A Review". *Journal of Environmental Management*. 302(114032). https://doi.org/10.1016/j.jenvman.2021.114032.

Warn, T. 2010. "Simcat 11.5 A Guide and Reference for Users". *Environment Agency (Available at: www.environmentagency.gov.uk/static/documents/.../111_07_SD06.pdf)*.

Wattel-Koekkoek, E., M. Van Vliet, L. Boumans, J. Ferreira, J. Spijker, and T. Van Leeuwen. 2012. "Soil Quality in the Netherlands for 2006–2010 and the Change Compared with 1993–1997: Results of the National Soil Quality Monitoring Network". *RIVM Report*. 680718003.

Wienhold, B. J., S. S. Andrews, and D. L. Karlen. 2004. "Soil Quality: A Review of the Science and Experiences in the USA". *Environmental Geochemistry and Health*. 26(2): 89–95. https://doi.org/10.1023/B:EGAH.0000039571.59640.3c.

Wienhold, B. J., D. L. Karlen, S. S. Andrews, and D. E. Stott. 2009. "Protocol for Indicator Scoring in the Soil Management Assessment Framework (SMAF)". *Renewable Agriculture and Food Systems*. 24(4): 260–66.

World Bank. 2016. *World Development Indicators Online*. Washington, DC.

World Bank. 2021. *World Development Indicators Online*. Washington, DC.

World Bank. 2022. *World Bank Country Classifications by Income Level*. 2023.

Yasmeen, R., I. U. H. Padda, X. Yao, W. U. H. Shah, and M. Hafeez. 2022a. "Agriculture, Forestry, and Environmental Sustainability: The Role of Institutions". *Environment, Development and Sustainability*. 24: 8722–46. https://doi.org/10.1007/s10668-021-01806-1.

Yasmeen, R., R. Tao, W. U. H. Shah, I. U. H. Padda, and C. Tang. 2022b. "The Nexuses between Carbon Emissions, Agriculture Production Efficiency, Research and Development, and Government Effectiveness: Evidence from Major Agriculture-Producing Countries". *Environmental Science and Pollution Research International*. 29(34): 52133–46. https://doi.org/10.1007/s11356-022-19431-4.

Yu, C., X. Huang, H. Chen, H. C. J. Godfray, J. S. Wright, J. W. Hall, P. Gong, S. Ni, S. Qiao, G. Huang, Y. Xiao, J. Zhang, Z. Feng, X. Ju, P. Ciais, N. C. Stenseth, D. O. Hessen, Z. Sun, L. Yu, W. Cai, H. Fu, X. Huang, C. Zhang, H. Liu, and J. Taylor. 2019. "Managing Nitrogen to Restore Water Quality in China". *Nature*. 567(7749): 516–20. https://doi.org/10.1038/s41586-019-1001-1.

Zhang, B., H. Tian, W. Ren, B. Tao, C. Lu, J. Yang, K. Banger, and S. Pan. 2016. "Methane Emissions from Global Rice Fields: Magnitude, Spatiotemporal Patterns, and Environmental Controls". *Global Biogeochemical Cycles*. 30(9): 1246–63. https://doi.org/10.1002/2016gb005381.

Zhang, L., R. E., M. M. Ali, H. Lin, S. Zhang, S. Jin, Z. Zhu, J. Hu, Y. Yao, Y. Sun, S. Yan, and Z. Liu. 2023. "Livestock and Poultry Manure Management from the Perspective of Carbon Neutrality in China F". *Frontiers of Agricultural Science and Engineering*. https://doi.org/10.15302/j-fase-2023509.

Zhang, W. 2018. "Global Pesticide Use: Profile, Trend, Cost/Benefit and More". In: *Proceedings of the International Academy of Ecology and Environmental Sciences*. Vol. 8. No. 1. 1.

Zhao, H., M. Yuan, M. Strokal, H. C. Wu, X. Liu, A. Murk, C. Kroeze, and R. Osinga. 2021. "Impacts of Nitrogen Pollution on Corals in the Context of Global Climate Change and Potential Strategies to Conserve Coral Reefs". *Science of The Total Environment*. 774: 145017. https://doi.org/10.1016/j.scitotenv.2021.145017.

Zilberman, D., J. Zhao, and A. Heiman. 2012. "Adoption versus Adaptation, with Emphasis on Climate Change". *Annual Review of Resource Economics*. 4(1): 27–53. https://doi.org/10.1146/annurev-resource-083110-115954.